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Effects of Fire on Madrean Province Ecosystems

A symposium proceedings

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Abstract: This second conference on the Madrean Archipelago/Sky Island ecosystem brought together scientists, managers, and resource specialists from government, universities, and private organizations in the United States and Mexico to explore the effects of fire on Madrean Province ecosystem, and how fire can be incorporated in an ecosystem approach to both research and management.

Keywords: Fire, ecosystem management, Madrean Archipelago, Madrean Province, Southwest

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Effects of Fire on Madrean Province Ecosystems

A symposium proceedings

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Opening Comments

Denver Burns [1]

On behalf of the Rocky Mountain Forest and Range Experiment Station, I welcome you to the second conference on the Madrean Archipelago. In just 18 months from the initial conference, you have convened this second conference to deal with issues of science and management of this region important to Mexico and the United States.

The first conference, Biodiversity and Management of the Madrean Archipelago: The Sky Islands of the Southwestern United States and Northwestern Mexico, was held here in Tucson September 19-23, 1994. That conference brought together researchers, managers, and private stakeholders to review current state-of-knowledge, discuss developing and achieving management goals, and identify gaps in knowledge and management needs. As pointed out by conference participants, many issues confronting both managers and researchers are common to both the United States and Mexico. Participants felt that the scientific basis for ecosystem management and maintenance of ecosystem viability in this region of exceptional biodiversity was not yet adequate. The roles of prescribed and natural fires and effects of fire on natural resources were identified as significant areas needing additional efforts. Participants also recognized differences in cultural values in the framework for conducting research and implementing management strategies between Mexico and the United States, and recommended increased interchange and collaboration. I can tell you that the pace of collaboration has increased in many areas, including inventory and monitoring.

This week's conference, Effects of Fire on Madrean Province Ecosystems, is designed as a significant step in the journey of meeting needs identified in the first conference; it is designed to continue development of an ecosystems approach in both research and management. This conference is again structured to bring together resource specialists, managers, scientists, and private stakeholders from both sides of the border to share experiences, opinions, and information needs on the effects of fire on resources and fire management in the ecosystems of the region. To continue enhanced collaboration across the border, the conference has been organized with leadership from Peter Ffolliott at the School of Renewable Natural Resources, University of Arizona, and Gilberto Solis-Garza of the Universidad de Sonora. Critical sponsorship and technical support are being provided by the Rocky Mountain Forest and Range Experiment Station, Coronado National Forest, and Southwestern Region of the USDA Forest Service, INIFAP, Centro de Investigaciones Biologicas del Noroeste, and Centro de Ecologia, among others.

This conference will expand on such collaborative efforts as the Sister Forest Agreement between the Coronado National Forest and el Estado de Sonora, which has fire management as a central theme.

These are challenging times for managing unique ecosystems to maintain and in some cases restore natural processes, to maintain and enhance biological diversity, and to sustain and improve productivity. We humans are an integral part of these ecosystems, and our needs must be addressed in maintaining ecosystem viability. Fire is a critically important tool and force in these efforts. The leadership in the use of fire coming from this part of North America will have important effects in other parts of the continent. Just last week, USA Today had a map showing that Arizona and New Mexico counties had population growth of at least 5% during the years 1990-1995. In fact, in the Interior West only a handfull of counties had less than 5% growth. Yes, population pressures will increase on our ecosystems and influence how fire might be used as a tool.

Each conference we have had with major participation by both Mexico and the United States has led to increased scientific knowledge, improved cooperation and coordination, and optimism that together we can understand the complexities of Madrean ecosystems and manage them for today and tomorrow. You have taken on the responsibility to continue the success of the previous conference.

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[1] *Director, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO*

Fire Management and Research Needs: Some Perspectives in the Madrean Province

Peter F. Ffolliott[1]

Abstract.—This conference provided a forum in which resource specialists, managers, researchers, and other interested people could share their collective experiences, opinions, and informational needs on 1) the effects of fire on the resources, and 2) fire management in the Madrean Province ecosystems. The conference presenters also updated the state-of-knowledge on these subjects. A group discussion toward the conclusion of the conference considered the perceived fire management and research needs in the Madrean Province ecosystems. The participants considered these needs in relation to ecosystem functioning, social-economic-cultural factors, prescribed burning programs, and environmental issues. A summary of this discussion is presented here.

INTRODUCTION

A conference in September 1994 brought together researchers, managers, and other interested people from government, university, and private sectors to examine the biological diversity and management challenges of the Madrean Province ecosystems in the southwestern United States and northern Mexico (DeBano et al. 1995). Topics considered included plant, vertebrate, and invertebrate ecology; hydrology, riparian systems, and aquatic resources; fire effects; conservation and management practices; human uses through time; and visions for the future. A discussion at the conclusion of that conference reviewed the then state-of-knowledge, and identified important gaps in knowledge that needed to be filled.

Among these gaps was insufficient knowledge on the role of fire in the region and, more particularly, its effects on the resources of the Madrean Province ecosystems. It was felt that the level of knowledge on the structure of vegetative communities, and the composition of vertebrate and invertebrate populations in response to fire frequency, were insufficient (DeBano and Ffolliott 1995). Furthermore, long-term consequences of large wildfires on natural and cultural resources were largely unknown.More study of the role of prescribed burning practices in reducing fuel accumulations and altering ecological habitats was also suggested. It was recommended, therefore, that a follow-up conference focusing on the effects of fire on natural and cultural resources of the Madrean Province be held in the spring of 1996. These proceedings represent the formal record of this second conference.

[1] *School of Renewable Natural Resources, University of Arizona, Tucson, Arizona.*

THE PURPOSE OF THIS CONFERENCE

This conference provided a forum in which resource specialists, managers, researchers, and other interested people could share their collective experiences, opinions, and informational needs on the effects of fire on the resources in the Madrean Province ecosystems and on fire management. The conference presenters also updated the state-of-knowledge on these subjects. A symposium held in 1988 (Krammes 1990) and a bibliography specifically prepared for this conference (DeBano and Ffolliott 1996) furnished a point-of-reference for this purpose.

Invited and poster papers covered a variety of topics including the ecological role of fire, historical perspectives on fire regimes, socio-economic perspectives, effects of fire and the management implications, and fire management in the region. Present-

ers came from both sides of the international border to share their knowledge.

A group discussion toward the conclusion of the conference considered perceived fire management and research needs in the Madrean Province ecosystems.2 Participants considered these needs in relation to ecosystem functioning, social-economic-cultural factors, planning prescribed burning programs, and environmental issues. This discussion is summarized here.

ECOSYSTEM FUNCTIONING

The participants felt that additional knowledge on the historical role of fire in the development and functioning of ecosystem communities in the Madrean Province would help establish a "benchmark" to identify how fire might accomplish specific management objectives. While much has already been learned in this regard, as witnessed by papers presented in the conference, more site-specific study was encouraged. While it was agreed that fire can meet a variety of management objectives in many cases, fire cannot meet all of the objectives of ecosystem management. The use of fire with other management practices (livestock grazing, silvicultural treatments, and wildlife habitat enhancement) is a likely scenario.

It was further stated that knowledge on the effects of fire on the development and functioning of riparian systems and wetlands in the Madrean Province is inadequate. Because of the increasing importance being placed on these often fragile ecosystems by the public (Shaw and Finch 1996, Tellman et al. 1993), increased study of the impact of fire on these systems is necessary. How fire alters the flow of water and sediments through the systems, and the subsequent changes in the development and sustainability of the characteristic biotic communities need to be determined.

It was also concluded that previous and planned fire response studies should be integrated to provide a more holistic picture on the effects of fire in Madrean ecosystems. At the same time, syntheses of fire responses at alternative scales of assessment are necessary. Assessments of responses at both landscape levels and individual plant levels are required. Knowledge of how fire affects vital components of ecosystems and the attributes of selected plant species is important. A review of information generated through efforts such as the International Biome Program might be a useful input to these assessments.

It is often thought that fire can be a managerial tool to restore ecosystems to past conditions. While it is true that fire can reverse successional patterns on many sites, the post-fire conditions obtained might not be acceptable to the public. The participants in this conference believed that it is important, therefore, that the past conditions desired be comprehensively defined by the public before fire is prescribed. An ecosystem is generally too variable for a single fire prescription to be totally effective, however. Furthermore, because natural change is to be expected, the use of fire in the Madrean Province must be carefully considered in a dynamic context.

The participants generally agreed that computer simulations and geographic information systems (GISs) in studying fire effects. Synthesizing simulation techniques from existing experiences, information, and databases can be helpful to fire managers in separating and then analyzing the effects of fire on ecosystem resources. Deficiencies in knowledge could also be recognized in the process of developing the simulations. GISs can be used to integrate databases on spatial scales to simulate the size and spread of fire, levels of fire severity, and the spatial effects of fire on ecosystem resources. It was suggested that the databases to be used in computer simulations and GISs be compiled in electronic formats for user accessibility and a better sharing of information.

SOCIAL-ECONOMIC-CULTURAL FACTORS

Historical perspectives might be helpful in changing public (social) perceptions of fire in situations where this change is crucial to obtaining necessary political and financial support. Fire specialists rarely have the political power or financial resources required to affect this change.

It was also concluded that long-term data sets on the benefits and costs of fire are prerequisite to effective economic analyses of burning programs. Dis-

[2] *The group discussion was moderated by Malchus B. Baker, Jr., Rocky Mountain Forest and Range Experiment Station, USDA Forest Service, Flagstaff, Arizona, and Duane A. Bennett, Coronado National Forest, USDA Forest Service, Hereford, Arizona.*

counting anticipated streams of benefits and costs to the present is necessary in making decisions on the economic feasibility of including fire in management situations. While available databases might serve as a foundation, these data are often too limited in scope for more thorough economic analyses. Unfortunately, there appears to be limited money available for research on the effects of prescribed burning and the implementation of large-scale burning programs at this time.

A number of issues involved in burning along the wildland-urban interface were the focus of a lengthy discussion among the participants. It was felt that more efficient land-use planning, zoning, and other legal regulations might become necessary to minimize the damage to cultural resources in this interface. Smoke easements, limitations on the density of housing and other buildings, and educating people on the immediate and long-term effects of fire represent other approaches that should be considered. The technique of Daniel and Ferguson (1991) in showing people pictures of possible post-fire futures was suggested as a method of helping the public to formulate their perceptions of fire in the wildland-urban interface. The problems of fire in these high-valued settings are expected to increase in the Madrean Province through time. It was recommended that the public and fire managers engaged in highly interactive discussions in addressing these problems.

Opportunities to use fire in beneficial ways to satisfy ecosystem management goals will depend largely upon public support. Cortner et al. (1984) indicated that public acceptance and understanding of the purposes and benefits of fire are high in the region, and that additional education is likely to increase the tolerance for fire. The participants in this conference also endorsed the call for increased education in fire effects at all levels through meetings, field trips, and simulations on both sides of the international border. People in Mexico often consider fire to be only negative in its impacts. Institutions with fire responsibilities should be encouraged to participate in these educational programs. Linked to the increased educational efforts should be expanded use of innovative technology-transfer mechanisms (such as the World Wide Web networks) whenever feasible.

PLANNING PRESCRIBED BURNING PROGRAMS

The participants agreed on the importance of planning at the ecosystem level, with public involvement, and within a holistic framework in prescribing burning programs. Differences between the public and responsible management personnel in their respective acceptability of prescribed burning as a tool to achieve management objectives must be reconciled to implement effective burning programs. The legality of burning along the wildland-urban interface must also be established in the planning process. While separation between fuel concentrations and cultural developments is declining in many instances, the need still exists to justify prescribed burning as a way to avoid major wildfire that is destructive to life and property.

More information on the characteristics of both prescribed fire and wildfire (including flame lengths, rates of spread, and severity) and post-fire effects on ecosystem resources is also necessary in planning prescribed burning programs. This information could be useful in quantifying the trade-offs between fire and resource values. The trade-offs between the use of fire and the resulting emissions must also be considered in the planning process.

Plant and animal resources should be inventoried in areas considered for burning. It is important that areas of high risk to these resources be delineated and, when warranted, avoided in the fire prescription. The participants suggested that computer simulations and GISs might be used to model the effects of burning on collective ecosystem resources. A common question asked by the public is whether the goals specified by management can be met by prescribed burning. Fire is not the best option for achieving management goals in all situations and, therefore, careful selection of areas for the implementation of prescribed burning programs is required.

ENVIRONMENTAL ISSUES

Much of what has been reported to this point relates to the environment. However, other environmental issues were also considered by the participants in this conference. It was felt, for example, that air quality, the level of visibility, and excessive smoke can be obstacles to the use of fire for management

purposes. Relatively little is known about these concerns in relation to prescribed fire vis-a-vis wildfire in the Madrean Province. Side-boards placed on the use of fire have been established on a large-scale by the Clean Air Act, with which the responsible management agencies must comply.

The participants also concluded that the comparative consequences of fire and long-term fire suppression on ecosystem properties have not been adequately studied in the region. Changes in species compositions, magnitudes of nutrient regimes, and activities of soil microorganisms are largely unknown. The degree to which these and other relevant ecosystem modifications are irreversible in both the short- and long-term should be determined to place fire in its proper and historical perspective.

The role of prescribed natural fire and its effect on ecosystem resources are generally unknown in the Madrean Province. Any plan to manage prescribed natural fire must consider a number of divergent issues (Bunnell 1995). These fires can be projected to be, and often become large events of long duration. The precise location, size, severity, and timing of their movements are largely unknown in the beginning stages of natural fire. The nature of these fire is ultimately defined by weather events. Some participants felt that the uncertainty of prescribed natural fire itself would limit its role in meeting and sustaining a desired future condition, although it was concluded that further investigation is warranted.

CONCLUDING COMMENTS

The topics summarized in this paper under the headings of ecosystem functioning, social-economic-cultural factors, planning prescribed burning practices, and environmental issues are not necessarily exclusive to those headings; in fact, many of these topics could be easily listed under several of the headings. The headings served largely as a facilitating framework for discussion.

It is also important to mention that the management and research needs presented in this paper are not meant to be all-inclusive. The discussion only reflects the dialogue among the participants at the conference. Importantly, no attempt has been made by the author of this paper, the discussion moderators, or the participants in the conference to prioritize the discussion points.

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Ecological Role of Fire in the Madrean Province

Larry S. Allen[1]

Abstract.—Fire has long been a significant factor in the ecology of the Madraen Uplands. The region is one of high lightning occurrence with frequent wildfires in spring and early summer. Primitive man is known to have used fire as a tool for hunting, warring, and manipulation of vegetation. Accidental fires have historically occurred in the area and they continue. With the advent of Europeans came modification of fuels by livestock grazing and other land management activities. This resulted in significant diminishing of the ability of fires to spread and affect large landscapes. As modern range management techniques, coupled with moderate stocking rates began to improve rangeland condition, several state and federal agencies initiated aggressive fire suppression. The result was virtual elimination of fire as a significant ecological force for almost a century. Elimination of fire from these ecosystems had far reaching ecological effects, and the landscapes that we see today are significantly different from those where fire has continued to be a factor. Landowners and agencies are currently exploring ways to restore fire to its natural role in the ecosystem.

INTRODUCTION

Landscapes of the Southwestern United States and Northern Mexico, including the Madrean Province have historically been shaped by a number of influences. One of the most widespread of these ecological factors has been fire (Allen 1994, Bahre 1991). Natural, or lightning caused fires are quite common throughout the province, with increasing frequency resulting from human activities.

Speakers at this conference will indicate that person- caused fires have influenced these ecosystems since long before European contact. It is widely believed that pre- Columbian native Americans used fire as a tool to manipulate vegetation, for hunting, and as a weapon of war (Spoerl this conference). Archeological sites often exhibit evidence of destruction by fire. However it is not usually possible to determine if such fires were accidental, arson, or intentional destruction upon abandonment. Modern inhabitants continue to use fire for the same purposes and are equally careless with this often beneficial tool.

Tree ring studies would indicate that the frequency of fires in forested ecosystems of the Southwest significantly declined late in the 19th century (Swetnam 1993). This change in frequency of natural burning is often attributed to suppression by state and federal agencies, but the timing of events suggests that other factors were involved. Early settlers were concerned about protection of property and forage from damaging fires, and they undoubtedly made some attempts at suppression. The first agency suppression program was initiated by the U.S. Forest Service as they assumed management of the National Forests about 1907. Early Forest Rangers were delegated management responsibility for entire mountain ranges. They had only primitive tools and traveled mostly by horse. If fires became too large for one or two firefighters, they were instructed to recruit help from ranchers, miners, loggers and other local residents. This low level suppression activity could not have resulted in the widespread exclusion of fire that is evident from that era. Swetnam (1993) noted that fire frequency declined in mountains in Sonora, Mexico at about the same time, in spite of the fact that no suppression was occurring there.

Significant ecological changes occurred from the late 1800's through the turn of the century. These

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changes have been attributed to a number of factors, including overgrazing and drought (Allen 1989, Bahre 1991, Hastings and Turner 1965). In most areas these changes resulted in an increase in woody plants at the expense of grasses and forbs, thus decreasing ground fuels available for the spread of wildfires.

Modification of fuels by man's activities, particularly grazing, combined with increasingly efficient fire suppression resulted in the virtual elimination of fire as an ecological force in the United States portion of the region. In many of the Madrean Mountains of Mexico, fires continue to burn in a more natural regime.

Recent changes in federal fire management policy recognize the beneficial role of fire as a tool for landscape management (USDI, USDA 1996). Perhaps the most important change in policy is the recognition that ecological factors must be a part of fire suppression decisions. This policy directs the Forest Service, Bureau of Land Management, National Park Service, and US Fish and Wildlife Service to plan for fire management on all federal lands, including an evaluation of the need for prescribed natural fire.

ECOLOGICAL INFLUENCES OF FIRE

Sonoran Desert

Sonoran desert ecosystems typically exhibit a great deal of bare soil at the time of year when fires usually occur, inhibiting the spread of ground fires. For this reason many species within these ecosystems have not developed defensive mechanisms and are quite susceptible to damage from fire. Inhabitants of the Sonoran Desert of Southern Arizona place great value on the giant sahauro cactus and other desert plants, and it has become desirable to protect these areas from fire at least in the vicinity of towns and along roads. Introduction of well-adapted exotic grasses, such as Lehmanns love grass and bufflegrass has added a hazard of fast spreading fires that was not historically present. This could increase the threat to valued desert plants and certain threatened, endangered, or rare species such as the Pima pineapple cactus.

Chihuahuan Desert

Chihuahuan desert plants also have evolved with limited fire in their environment, and many are susceptible to fire. Although the plants of this desert are generally not as spectacular as those in the nearby Sonoran zone; many yuccas, sotols, and cacti add variety and beauty to an otherwise drab landscape making protection from damaging fires appropriate. On these typically hot, dry sites black grama grass is thought to be particularly susceptible to fire. This grass is valuable for livestock and wildlife forage, cover for birds and small mammals, and watershed protection. Exotic grasses are not as common in the Chihuahuan desert as in the Sonoran, and few sites will carry a large fire.

Semi-Desert Grassland

Adjacent to most desert areas of the Madrean Province are semi-desert grasslands. These areas are characterized by a mixture of grasses and forbs with scattered shrubs and trees. Typical desert plants will invade these grasslands in response to site disturbance or drought. Common plants encroaching on these grasslands include mesquite, catclaw, sandpaper bush, algerita, agaves, yuccas, cacti, snakeweed, burroweed, turpentine bush, and baccarus. Historically these grasslands have been maintained in an open aspect by frequent wildfire (Wright 1980, Humphrey 1958). As livestock were introduced as early as the 1700's and numbers peaked about 1890, two significant ecological factors came into play. Heavy grazing removed competing grasses and forbs, creating a favorable microclimate for shrubs, cacti and succulents. At the same time ground fuels were removed by livestock and drought, and fires were no longer able to spread as they had historically (Allen 1994).

As grazing management improved, agency suppression became effective to the point that most fires are suppressed while still quite small. This has resulted in interruption of the ecological role of fire and replacement of many former grasslands with desert scrub vegetation. Once the grassland aspect is lost, it is exceedingly difficult to restore these sites. Most agencies and landowners see a value in restoring fire as a natural ecological force, but on many sites some form of shrub control (either mechanical or chemical) will be required to produce enough fuel to carry a fire. Once ground fuel continuity is reestablished, it should be possible to maintain the grasslands with some combination of natural and management ignited fire.

Plains Grassland

On slightly more mesic sites of the Madrean Region are areas of Plains Grasslands with floral affinities to the Great Plains of the central United States. Like the semi-desert grasslands, these areas are susceptible to an incursion by shrubs and trees. Common invading species include pinyon pine, junipers, several species of oak, skunkbush, snakeweed, and some semi-desert plants such as mesquite and catclaw (Dwyer & Piper 1967). The plains grassland was historically maintained by fire and grazing by large herbivores. The same factors described for the semi-desert also resulted in exclusion of fire from the plains grassland.

Efforts to reestablish fire to this ecosystem often involve deferment of grazing to create fuel followed by controlled burning. In many cases a combination of moderate grazing and a fire suppression policy which considers possible benefits from wildfire will restore this grassland.

Montane Grassland

At the higher elevations in the Sky Islands are occasional mountain meadows and other montane grasslands. These grassy openings in the coniferous forest have historically been maintained by fire, and exclusion of this ecological force has often resulted in encroachment of conifers. In the 1960's and early '70's the Forest Service occasionally maintained these grasslands with hand thinning. Increasing labor costs coupled with concerns for sensitive species have inhibited efforts of the agency to maintain this grassland. Ground fuels are generally adequate to carry a fire, if suppression practices were modified.

Madrean Oak Woodland

At intermediate elevations of the Madrean Province are extensive areas of evergreen oak woodlands (Brown 1982). These areas typically exhibit a mosaic of savannahs, dense woodlands, and grassy openings. This mosaic is often influenced greatly by aspect and topography, with ridge tops and south facing slopes supporting a more open stand of oak with abundant grass in the under story and interspaces. Predominant species include Emory and Mexican blue as well as some Arizona white, Toumey, and silverleaf oaks. Scattered pinyon, juniper, and mesquite trees are common. This ecosystem is one of the most productive for livestock products and wildlife habitats in the Southwest and productivity is dependent on maintaining at least part of the area in more open woodlands (Allen 1994). Fire has been effectively excluded from this ecosystem by agency suppression efforts.

Mature evergreen oaks are not particularly susceptible to fire (unless under drought stress) but fire is an effective tool in preventing establishment of new seedlings. In general ground fuels are sufficient to provide for fire spread. Use of fire as a management tool will sometimes require deferment of grazing.

Pinyon Juniper Woodland

Not as extensive as the oak woodland but also significant in the Sky Islands is the Pinyon Juniper Woodland. Pinyon pine, and several species of juniper are the prominent species of this woodland. Many evergreen oaks species are also present These woodlands are typically less productive of herbaceous vegetation than those dominated by oaks and therefore more easily modified by site disturbances such as grazing. These sites are often dominated by trees and shrubs with little ground fuel. This lack of fuel continuity coupled with agency suppression has effectively eliminated fire as an ecological factor (Dwyer & Pieper 1967). Fire is effective in controlling most woody species in the pinyon juniper woodland, with the notable exception of alligator juniper and mesquite.

Desired vegetation for these sites usually involves a less dense woodland, with savannahs and grassy openings scattered throughout the woodland. This will often require mechanical or chemical plant control, and deferment of grazing before fire can be used as a tool or reestablished as an ecological force.

Riparian Areas

Many stream side areas support riparian vegetation such as cottonwood, sycamore and willow trees, and deergrass. These areas have great value to the many neotropical birds found in the area as well as most species of resident wildlife. Shade, green vegetation, and cooler temperatures make these areas very attractive to livestock and human recreationists, thus concentrating use on about 2 percent of the

acreage. Because of these exceptional values, it is important that fire not be allowed to damage the vegetation. Like the surrounding uplands, southwestern riparian areas evolved in an era of frequent wildfires, and natural fires are unlikely to do significant damage in these areas except under unusually severe conditions. Prescriptions for management ignited fires must provide for a relatively cool burn in riparian areas.

Pine Oak Forest

At the transition between the oak woodland and higher elevation conifer forests is the pine oak forest. This forest type typifies the intermediate elevations of Northern Mexico. Like other Madraen ecosystems these forests evolved with frequent wildfires. Since most of these areas are in Mexico, they have experienced a less intense suppression effort than would have been the case in the US. As Mexican foresters become better trained and equipped to control conflagrations, there is a danger that they will repeat the mistakes of their neighbors to the North and exclude fire from the forest. Finding a balance between the need to control damaging fires and the beneficial role of fire in the ecosystem is a significant challenge facing Mexico today.

Prescribed fire and some form of managed wildfire will be required in the management of the pine oak forest.

Pine Forest

The most valuable commercial forest in the southwestern US and Northern Mexico is the pine type. In the US portion of the Madrean Province these forests are made up of ponderosa, southwestern white, Apache, Arizona, and Chihuahua pines with some Douglasfir and true firs. Alligator juniper and Gambel oak are also common components of the forest. In Mexico, pine forests have a similar aspect but include many other species. Forty two species of pine are known from Mexico (SARH 1996).

Exclusion of fire from pine forests of the United States by over-aggressive suppression efforts of the Forest Service and other agencies has created a situation where unnatural fuel accumulations threaten catastrophic wildfires throughout the west. The severe fire season of 1994 was a precursor of things to come in these overstocked forests (Allen 1994). The Rattlesnake Fire in Arizona's Chiricahua Mountains is one example of the result of years of over protection.

Lightning ignited a fir near Rattlesnake Peak during the severe dry period of the summer of 1994. A prompt initial attack by a Forest Service helitack crew failed to bring the fire under control and additional forces were dispatched. The point of origin was in a pine forest, with abundant ground fuels and a multi-storied stand of pines and other conifers. The multi-storied nature of the stands created "fuel laddering," which allows ground fires to climb to the tree crowns. A combination of extreme weather conditions, dangerous fuel arrangements, and limited resources due to numerous other fires in the zone and around the West made it impossible to safely confine the fire with crews available the first day. The fire spread quickly uphill toward the mixed conifer and spruce areas of Rustler Park and the Chiricahua Wilderness.

In spite of a 6.5 million-dollar expenditure, the Rattlesnake Fire resisted all suppression efforts for about three weeks, and eventually burned 27,500 acres. Severity of the burn varied with slope, aspect, fuel accumulations, and vegetation type. Pine forests which lacked laddering fuels were lightly burned, with the effect of sanitation of the forest floor and prevention of excessive fuel buildups. Ninety years of aggressive fire suppression had resulted in growth of many dense thickets of pine saplings. Where the fire burned through these "jack pine" thickets, many sites were reduced to blackened stumps, with all ground cover removed. Summer thunderstorms immediately following the fire resulted in significant soil loss from these sites. Damage to these watersheds is severe and long term, but the healing process has begun with native grasses, forbs, and shrubs beginning to pioneer the site.

Mixed Conifer/Spruce

At the summit of the Sky Island ranges is an extensive mixed conifer forest, with limited areas of spruce and spruce-fir. These forests occur on cool, most moist sites and they tend to grow in dense stands with little understory. Soils are protected by a typically deep layer of litter on the forest floor. In a natural fire regime frequent ground fires would open the stands and remove the accumulation of down material from windthrow and other natural causes

(Swetnam 1993). Exclusion of fire from this ecosystem for almost a century has resulted in an unnatural buildup of fuels on many Madrean mountain tops. Fuel accumulations of 200 to 300 tons per acre are common in these forests in the U.S. Significant areas of the Chiricahua Wilderness had 200 or more tons per acre prior to the Rattlesnake Fire.

In general the mixed conifer did not burn as hotly as the pine in the Chiricahuas, but some areas with heavy ground fuel accumulations were severely damaged. Down logs and natural slash created a heat buildup that resulted in mortality of large mature trees. Because these mixed conifer forests have little or no understory the watersheds became significantly at risk for accelerated erosion. Rucker Lake, a small recreational lake downstream from the fire, was completely filled with silt by fall. Picnic tables at the lake were buried. Hydrologists caution against any investment in restoring the lake and campgrounds until the watersheds have had two or three years to stabilize.

Fire intensity in the spruce and spruce-fir within the Rattlesnake Burn was much less severe than other conifer forests, with little or no damage to the forest or the watershed. Because the mixed conifer and spruce-fir plant communities occur on cool moist sites, severe fire years are uncommon and these are sometimes referred to as "asbestos forests." When the inevitable bad years do occur, fires in this fuel type will be extremely difficult to control, due to current fuel loading, and damage will be severe. Fire season of 1994 demonstrated this throughout the west.

The Rattlesnake Fire occurred mostly in a wilderness area and adjacent undeveloped areas. Firefighters were able to save all structures in its path. Many areas with similar fuel buildups include summer homes, organization camps, developed recreation sites, electronic sites, telescopes, and even communities. The Pineleno Mountains, to the north, have a more severe fuel accumulation than that found in the Rattlesnake Fire and a great deal of human occupancy.

In areas like the Pinelenos it will not be possible to restore a more natural fire regime, without first making significant modifications to existing fuel concentrations. Prescribed fire would be one tool of choice, but this would first require creation of fuel breaks and reduction of concentrations through mechanical means. In many cases an opportunity exists to do fuels reduction through sale of various forest products. Attempts to manage these fuels are complicated by a number of factors including the presence of a number of species with affinities for old growth, or dense forests; a segment of the public that will resist any silvicultural solutions through appeals and litigation; lack of a local forest products industry; and bureaucratic inertia of many involved agencies.

CONCLUSIONS

- Fire has long been a significant ecological factor in shaping Southwestern ecosystems.
- Several factors brought about an interruption in historic fire frequencies in the Maderean Province beginning in the late 1800's and continuing to the present time.
- A new federal fire policy directs agencies of the Departments of Agriculture and Interior to make efforts to "restore fire to its natural role" and to consider ecological impacts and benefits in fire suppression decisions.
- Fire managers must realize that desert plants are valued by our society and fire has the potential to do damage in the Sonoran and Chihuahuan Deserts. Planning and suppression decisions must reflect these values.
- Most agencies and landowners understand the value of fire as a tool and a natural ecological force.
- In the woodlands and grasslands of intermediate elevations of the region, most fires are beneficial and consideration should be given to allowing more area to burn, except where a threat to life or property is perceived.
- Unnatural fuel buildups will challenge managers' ability to restore fire to forested ecosystems of the province.
- As land managers in Mexico develop their ability to manage and suppress fires, they should be careful not to repeat the mistakes of the United States. Fire planning must recognize the beneficial role of fire in most ecosystems of Northern Mexico.

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The Role of Fire in the Southwestern Borderlands Ecosystem Research Program

Carleton B. Edminster[1]

Abstract.—The Southwestern Borderlands Program is the primary ecosystem management research program in the Malpai Borderlands area of the Madrean Archipelago sponsored by the Rocky Mountain Forest and Range Experiment Station. Fire, either as a natural or managed process, is a key component of the research program being conducted through cooperative efforts with research and management partners.

INTRODUCTION

The Southwestern Borderlands Ecosystem Research Program is one of several research efforts chartered in 1994 through the ecosystem management research initiative of USDA Forest Service Research. I wish to extend special recognition to my predecessor, Leonard DeBano, retired from the Rocky Mountain Forest and Range Experiment Station, and Larry Allen, Coronado National Forest, for their efforts in developing the successful initial proposal, unit structure, and research and management collaboration. The support of the Malpai Borderlands Group (McDonald 1995) and the Animas Foundation at Gray Ranch, New Mexico, continues to be a major contribution to the success of the effort as well. Gerald Gottfried, also of the Rocky Mountain Station, has been instrumental in assisting with coordinating the research effort.

The primary geographic scope of the program is Chihuahuan desert and semi-arid grasslands, shrub lands, and forested mountain ranges in southeastern Arizona and southwestern New Mexico. Results from the project are generally applicable to the Madrean Archipelago biogeographic province including northern portions of the states of Chihuahua and Sonora, Mexico. The Borderlands area is a unique, relatively unfragmented landscape of approximately one million acres containing exceptional biogeographic diversity in a series of communities ranging from desert grasslands to mixed conifer forests. The project focus area includes the San Bernardino Valley, southern San Simon Valley, Peloncillo Mountains, Animas Valley, Animas Mountains, and east into the Playas Valley. The project area has also been selected by the Coronado National Forest and Natural Resources Conservation Service as an area for implementing and demonstrating ecosystem management strategies.

THE BORDERLANDS RESEARCH PROGRAM

The mission of the program is to contribute to the scientific basis for developing and implementing a comprehensive ecosystem management plan to restore and maintain natural processes and improve and sustain the health and productivity of grasslands and tree lands. Maintaining the health and productivity of these natural communities is of critical importance in maintaining viable rural economies to sustain open landscapes. To the extent possible, much of the research program has been structured based on input from the earlier Sky Islands symposium (DeBano et al. 1995).

Within the Borderlands area, fire may be viewed as a vital natural process (Wright and Bailey 1982, McPherson 1995, Swetnam and Baisan 1996). During the last century, wildfire suppression and exclusion, past overgrazing, and recent droughts have had major impacts on the area's grasslands, desert grass-shrub, and forest communities (Bahre 1991, Allen 1996,

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McPherson and Weltzin 1997, Swetnam and Baisan 1996). Most notable have been changes in fire frequency and in the case of forests, severity. Prescribed natural and human-ignited burning are being developed as primary ecosystem management tools to reduce fuel accumulations and the incidence of catastrophic fires, improve plant community composition and forage condition and production, and improve wildlife habitat (Bahre 1991, Allen 1995). However, degraded desert grass-shrublands and forests with high fuel accumulations are delicate systems which often cannot be managed with fire without first being restored using other cultural reclamation techniques (Wright and Bailey 1982). The knowledge base for applying fire management in these conditions is generally incomplete (Ffolliott 1996). The changing attitudes and revised agency policies on natural and human-ignited prescribed fire that provided a major catalyst in the formation of the Malpai Borderlands Group (McDonald 1996) are a source of optimism for the future.

SPECIFIC STUDIES WITH A FIRE EMPHASIS

The strategy for accomplishing the research program mission has been to develop comprehensive multi-disciplinary syntheses of the status of our knowledge, identify research and monitoring needs and priorities, and define and implement critical studies and field experimentation. In much the same way that fire is viewed as a vital natural process, a major portion of the research program is dedicated to increased understanding of the natural role of fire and fire as a management tool. The following summary is limited to studies with a fire emphasis. Results thus far for many of these studies are contained in other presentations at this symposium, and the audience/reader is directed to these papers for additional detail.

The three major efforts which synthesize information and are nearing completion are:

- The status-of-knowledge on the role and importance of human and natural disturbances on Borderlands plant communities, prepared by Guy McPherson and Jake Weltzin of the School of Renewable Natural Resources, University of Arizona (McPherson and Weltzin 1997). The primary focus for this review is on the physiognomic changes in grasslands and woodlands. Fire and livestock grazing are historically the dominant disturbances. Seasonal shifts in precipitation and potential climate change are viewed as critical factors in looking to the future. The need for designed experiments in testing hypotheses is emphasized.
- Review of prehistory and early history of the Borderlands ecosystem developed by Paul Fish and Suzanne Fish of the Arizona State Museum (Fish 1996). A focus of the review is relationship of human ecology to the environment with fire use as a significant issue.
- Development of a comprehensive bibliography for the northern Madrean biogeographic province by Peter Ffolliott of the School of Renewable Natural Resources, University of Arizona. A sample of the fire-related section of the bibliography is included with registration materials for this symposium.

Current studies implemented to meet critical research and monitoring needs include:

- Development of a southwestern regional perspective of fire regimes and stand structure demography along elevational gradients from pinyon-juniper through ponderosa pine to mixed-conifer and spruce-fir forests by Thomas Swetnam and Christopher Baisan of the Laboratory of Tree-Ring Research, University of Arizona (Swetnam and Baisan 1996). This study builds on past site-specific investigations and includes recent efforts in the Huachuca Mountains (Danzer et al. 1996) and the Chiricahua Mountains (Kaib et al. 1996, Seklecki et al. 1996).
- Use of fire regime reconstruction in gallery pine-oak forests including frequency, seasonality, and area extent to infer fire frequency in adjacent grasslands by Mark Kaib and Thomas Swetnam (Kaib et al. 1996). Preliminary results suggest fire intervals in grasslands between 4 and 8 years.
- Investigation of fire frequency on nutrient budgets of grasslands, including differences in soil, plant cover, and nutrients for 3 fire frequencies over a 20-year period at Fort Huachuca, by Robert Webb of the Desert Laboratory of the US Geological Survey and Thomas Biggs and Jay Quade of the Department of Geosciences (Biggs et al. 1996). Preliminary results suggest a fire frequency not exceeding 2 fires per decade may be advantageous in control-

ling mesquite; more frequent burning may deplete soil macronutrients.

- Understanding and modeling the spatial pattern of fire regimes, fire habitats (including terrain analysis) along with fuels structure, fire behavior, and fire effects at landscape scales, by Stephen Yool, Michael Medler, Mark Patterson, and John Rogan of the Department of Geography and Regional Development, University of Arizona (Medler et al. 1996). Study areas include the Coffeepot Fire in the San Mateo Mountains on the Cibola National Forest, the Rattlesnake Fire in the Chiricahua Mountains, and the Baker prescribed burn in the Peloncillo Mountains. Studies involve remote sensing and spatial analysis techniques in fire prone terrain, along with extensive ground surveys.
- Monitoring the effects of prescribed fire on birds and vegetation at the landscape scale in the Baker Burn area in the southern Peloncillo Mountains, including the interactions of nectar-feeding bats with Palmer agave, by Peter Scott of the Department of Life Sciences, Indiana State University. This study, along with the above spatial pattern study, demonstrates the generally beneficial effects of a typical mosaic pattern resulting from prescribed burning in the grass-shrub type.
- Monitoring the effects of land-use history and resulting historical landscape change through the use of repeat photography, by Raymond Turner, retired, and Robert Webb. The effects of fire suppression and increased density of shrubs are documented.

DISCUSSION

The above listing of fire-related studies supported by the Southwestern Borderlands Ecosystem Research Program represents only what has been feasible given available resources. An ecosystem approach requires many research and management partners working in collaboration. The ecosystems in the Borderlands area are complex and dynamic. Additional involvement from other partners which contributes to the overall mission is always welcomed. A challenge is to facilitate productive working relationships to meet mutual goals and adapt the research effort to meet the needs of management and science. The next major prescribed burn planned for the Peloncillo Mountains is the Maverick Burn in an area north of the Baker Burn. Once the burn is scheduled, additional studies will be implemented to investigate the spatial pattern of fire effects on vegetation and wildlife. These prescribed burns are experiments in ecosystem management and represent unique opportunities to study the effects of fire as a management tool.

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Fire Histories of Montane Forests in the Madrean Borderlands[1]

Thomas W. Swetnam and Christopher H. Baisan[2]

Abstract.—In this paper we summarize historical fire regime patterns reconstructed using fire-scarred tree-ring specimens from seventeen montane forest sites in the Madrean Borderlands. In addition to a brief description of general patterns we also illustrate, with examples, several unique fire occurrence patterns influenced by land-use history and landscape configurations. Mean fire intervals and other statistical descriptors of fire interval distributions show that widespread surface fires were frequent in nearly all forests before ca. 1900, with fires occurring at least once per decade. Most fires occurred during the arid foresummer and lightning fire season from April through June. High spatial and temporal variability of fire frequency and other fire regime properties point to the importance of unique site and time-specific factors in controlling fire occurrence. These factors include the continuity of fuels and topography, livestock grazing, and possibly, the burning of some areas by Apaches during certain time periods. Ponderosa pine and mixed-conifer stands in rugged mountain ranges, such as the Animas in New Mexico, sustained mixed-fire regimes of both surface and crown fires. Frequent, widespread, surface fires ceased to occur in most U.S. Borderland sites at about the time intensive livestock grazing began. In contrast, montane forests on the Mexican side of the border sustained continuous surface fire regimes throughout the 20th century. With the elimination of frequent, widespread surface fires on the U.S. side, woody fuels have greatly increased in amount and continuity, and as a consequence, the size and intensity of recent crown fires were probably historically and ecologically anomalous.

INTRODUCTION

Wildfires can be an awesome force in the dynamics of Madrean forests and woodlands. This fact was brought close to home in southern Arizona in 1994 and 1995 with the sight of mushroom clouds of smoke rising over the Santa Catalina and Rincon Mountains, just a few kilometers away from Tucson. More than 33,000 hectares were burned by intense surface and crown fires during those summers (Allen 1995). In an earlier era these fires would have been viewed almost universally as purely destructive. However, we have learned from abundant historical and ecological research that fire is not always an enemy of forests, woodlands, and grasslands. On the contrary, because of evolutionary adaptation over millions of years, many Madrean plant and animal species are resistant to fire, or depend upon fire for maintaining and sustaining presence and abundance within their biotic communities (Barton 1993, 1994, Brown 1982, 1994, Caprio and Zwolinski 1995, Humphreys 1984, Marshall 1963, McLaughlin and Bowers 1982, Rogers and Steele 1980, Wright and Bailey 1982, Zwolinski, this volume).

Still, we have cause for concern. The problem is the amount and magnitude of ecological change that humans have caused during the past century. The most important of these changes is the nearly complete elimination of frequent, low to moderate intensity surface fires that used to burn over enormous areas. With the cessation of episodic fires, woody fuels (both living and dead) have accumulated across Southwestern landscapes. These accumulations now fuel explosive wildfires of intensities and sizes that these areas have probably not sustained for many centuries, if ever.

[1] This research was supported in part by funds provided by the Rocky Mountain Forest and Range Experiment Sation, Forest Service, U.S. Department of Agriculture.

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This situation creates a dilemma in our management of fire and landscapes. Fire is needed because certain plants and animals depend upon it, and because the accumulating fuels cannot be practically eliminated by any other means. Thus, we recognize that the fire process must be re-introduced in some areas. But by allowing lightning-ignited fires to burn, or by setting prescribed fires, we risk creating historically and ecologically anomalous patterns if these fires burn more intensely and in larger patches than biotic communities have previously sustained and are adapted to. Even more worrisome, we also risk the loss of human lives and property, especially with the current proliferation of homes and other structures (e.g., telescopes) within these flammable landscapes. On the other hand, even if we continue to aggressively attack fires with all means available, we will continue to have increasingly dangerous and destructive wildfires as long as fuel accumulations are allowed to persist and build. For example, studies of trends in fire statistics in the greater Southwest indicate that large, high intensity fires are increasing in frequency (Covington and Moore 1994, Sackett and others 1994, Swetnam 1990).

One of the main purposes of our research program is to place current fire regimes into historical context. Reconstructed fire histories (from tree-rings in this case) provide a baseline, or set of reference conditions, for understanding past fire regimes. These histories provide hard data and evidence for comparisons between current and past fire regime patterns. These comparisons tell us when, where, and how much fire regimes have changed, and help us determine if current patterns are unprecedented or unusual over the long term (i.e., historically or ecologically anomalous). In some circumstances, such unprecedented conditions are unlikely to be sustainable. We agree with Kaufmann and others (1994) and others (e.g., Allen 1994, Morgan and others 1994, Swanson and others 1994), that knowledge of historical-ecological "reference conditions" (or range of historical or "natural" variability) is essential for informed, science-based ecosystem management planning and decisions.

This recognition of the value and uses of reference conditions is not an attempt to restore some kind of mythical, pristine past. In some circumstances, historically or ecologically anomalous conditions may not be cause for concern, or call for special management actions. This depends upon a number of factors, such as:

1. The management objectives for the particular landscape in question,
2. How extreme the ecosystem changes are relative to the pertinent historical reference period,
3. Whether the new conditions are a detriment to sustainability of desired biota or other resources (e.g., watershed values or soils), and
4. Whether these changes are evidently caused by past human actions that presumably can be modified (e.g., total fire suppression efforts), or by natural forces beyond human control (e.g., unusual weather events, etc.).

Furthermore, reference conditions do not necessarily establish a template for management targets, or desired future conditions. Again, this depends upon the land being managed and the management objectives established for it. For example, reference conditions may be more directly relevant and applicable for setting restoration targets or goals in designated wilderness, parks, and natural areas than for other types of public lands. However, many of these conditions may be consistent with objectives and goals of restoring and sustaining productivity and biodiversity on other public and private lands as well. Although the historical record may not provide us with specific management targets, or tell us precisely how to restore desired conditions, it does provide a key element that is needed for understanding current ecosystems and landscapes: *historical perspective*. By understanding how ecosystems operated in the past, and knowing how we arrived at current conditions, we will be much better informed and prepared to restore and manage these ecosystems in the future.

The purpose of this paper is to provide a historical perspective of past fire regimes in the Madrean Borderlands. Over the past 15 years we (the authors, our students and collaborators) have conducted many different tree-ring based fire history studies in the montane forests and woodlands of this region. (Additional descriptions and comparisons of the range and variability of pre-settlement fire regimes of these Borderlands sites and more than 40 other sites in Arizona and New Mexico are described in Swetnam and Baisan [1996]). In this paper we lists the locations and characteristics of study sites, and we describe

fire interval statistics for each site. Each of these fire history reconstructions is a unique narrative of fire events that occurred in a specific location over periods of at least two centuries. Hence, each history offers a somewhat different insight on the role of landscape patterns (vegetation, elevation, topography), land-use history, and climate in controlling past fire regimes. Since we do not have the space here to describe all of the sites in detail, we have chosen a set of three different study areas that exemplify both general and specific patterns we have observed in the Madrean Borderlands. These case histories serve as a vehicle to point out the influence of land-use history and topography on fire regimes.

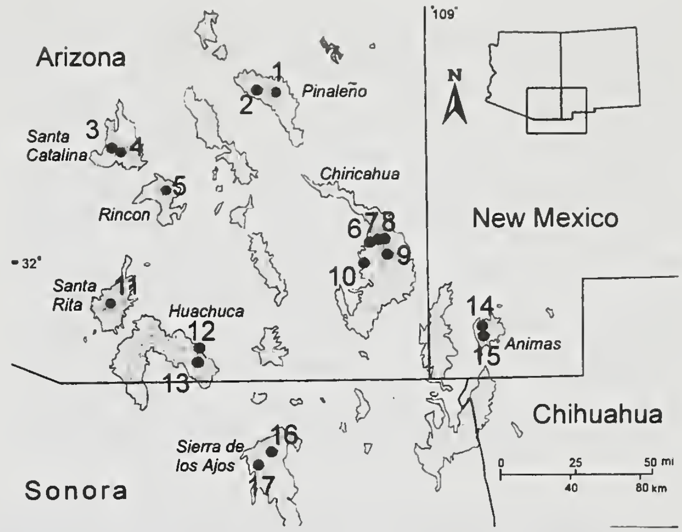

Figure 1. Map of the Borderlands area of southern Arizona and New Mexico and northern Sonora and Chihuahua, with woodlands and montane forests shown by shaded areas, and fire history study sites shown by dots. The numbers adjacent to the dots refer to sites listed in Table 1.

STUDY AREAS

Our fire history study sites are located within the montane forests of the "sky island" mountains (Fig. 1). The sites range in elevation from about 1,700 to 2,900 meters, and are located in a variety of topographic settings and forest vegetation types (Table 1). More detailed descriptions of these fire history study sites and findings are contained in the cited references in Table 1 (see papers in this volume by Danzer and others, Kaib and others, and Seklecki and others). In general, sites are located in montane forests within canyons, on ridges, saddles, or slopes. We have classified the forest vegetation in these sites into three types:

1. **Pine/Oak** — The overstory is dominated by lower elevation pines, such as Chihuahua pine (*Pinus lieophylla*) and Apache pine (*P. engelmannii*), but also ponderosa pine (*P. ponderosa*). Various mixtures of Madrean oaks (*Quercus spp.*), Arizona madrone (*Arbutus arizonica*), pinyon (*P. discolor*), junipers (*Juniperus spp.*), and various shrubs are sometimes co-dominant, or present in the understory of these pine-dominated stands.

2. **PIPO** — These are stands dominated by ponderosa pine. Some stands have incidental occurrence of other tree species (such as found in the PIPO/MC class described below), but in general these are pure or nearly pure ponderosa pine stands.

3. **PIPO/MC** — These are mixed-conifer stands, but ponderosa pine is the largest component. Other co-dominant tree species may include Douglas-fir (*Pseudotsuga menziesii*), southwestern white pine (*P. strobiformis*), and white fir (*Abies concolor*). In the pre-fire suppression era, many of these stands were probably pure or nearly pure ponderosa pine stands, but with fire suppression the more shade-tolerant, less fire resistant Douglas-fir and white fir have increased in numbers.

4. **MC** — These are mixed conifer stands where Douglas-fir and white fir are the largest components, but ponderosa and/or southwestern white pine are present as smaller components.

Since the fire histories are based upon tree-ring analyses of fire-scarred trees, we are limited to vegetation zones where woody trees produce clearly defined annual rings. This restriction has prevented our direct reconstruction of past fire regimes in lower deserts, grasslands, and woodland savannas (however, see Kaib and others, this volume.) Unfortunately, most of the woodland tree species in southern Arizona, New Mexico, and northern Mexico are not (yet) useful for accurate tree-ring analyses because growth rings are very indistinct, and in some cases, are probably not annual. These species include most of the evergreen and deciduous oaks (*Quercus spp.*),

except *Q. gambelli*, which is generally found at elevations above the woodlands. Because of numerous false rings (intra-annual latewood bands), none of the Borderlands junipers (e.g., *Juniperus monosperma, J. deppeana*) have been reliably dated, and Arizona cypress (*Cupressus arizonica*) has been used only in two studies (Moir 1982, Swetnam and others 1989). Mesquite (*Prosopsis glandulosa*) growing in relatively mesic sites has shown some promise for dendrochronological applications (Flinn and others 1994). Several riparian species in the Borderlands may be useful in future fire history studies (e.g., Arizona sycamore [*Platanus wrightii*], Fremont cottonwood [*Populus fremontii*]), but fire scars on these trees are rare and typically associated with such extensive heartrot that useful specimens are difficult to find.

METHODS

Site Selection Strategies

The study sites were selected in montane forest and woodland types and landscape situations that were common and thought to be generally representative of the management units where the studies

Table 1. Fire-scar study site names and descriptions. Map numbers in parenthesis next to site names refer to locations in Figure 1. See text for explanation of forest type abbreviations.

Site name (Map no.) Mountain range	Forest type	Min. elev (m)	Max. elev. (m)	Number of trees sampled	Inner ring date	Outer ring date	Reference
Rhyolite Lower (6) Chiricahua	Pine/Oak	1707	1804	12	1466	1987	Swetnam and others 1989; 1992
Pine Canyon (10) Chiricahua	Pine/Oak	1707	1829	27	1540	1995	Kaib and others, this volume, Kaib, in prep.
Rhyolite Middle (7) Chiricahua	PIPO/MC	1804	1920	30	1466	1987	Swetnam and others 1989, 1992
Sierra Ajos Ridge (16) Sierra Ajos	Pine/Oak	1981	2073	13	1438	1989	this paper
Sierra Ajos Saddle (17) Sierra Ajos	Pine/Oak	2100	2100	12	1438	1989	Dieterich 1983, Baisan and Swetnam 1995
Rhyolite Upper (8) Chiricahua	PIPO/MC	2073	2134	16	1466	1987	Swetnam and others 1989, 1992
Josephine Saddle (11) Santa Rita	Pine/Oak	2073	2195	17	1452	1979	Ortloff and others 1995
Sawmill Canyon (12) Huachuca	Pine/Oak	2012	2225	23	1499	1994	Danzer and others, this volume, Danzer, in preparation
Rose Canyon (4) Santa Catalina	PIPO	2134	2316	11	1558	1986	this paper
Animas North (14) Animas	PIPO	2438	2438	18	1538	1992	Baisan and Swetnam 1995
Animas South (15) Animas	PIPO/MC	2438	2438	56	1445	1992	Baisan and Swetnam 1995
Rustler Park (9) Chiricahua	PIPO/MC	2438	2591	58	1614	1995	Seklecki and others, this volume
Mica Mountain (5) Rincon	PIPO/MC	2070	2600	44	1481	1983	Baisan and Swetnam 1990, Baisan 1990
Pat Scott Peak (13) Huachuca	PIPO/MC	2545	2652	34	1499	1994	Danzer and others, this vol., Danzer, in prep.
Lemmon Peak (3) Santa Catalina	MC	2667	2731	16	1597	1920	this paper
Peter's Flat (2) Pinaleno	MC	2804	2880	40	1376	1993	Grissino-Mayer and others 1995
Camp Point (1) Pinaleno	MC	2900	2926	50	1543	1993	Grissino-Mayer and others 1995

were conducted (e.g., National Forests and National Parks). These were also areas where it was possible to carry out fire history reconstructions because the required tree-ring materials were present (i.e., abundant and well-preserved, living and dead fire-scarred tree specimens). Theoretically, such subjective site selection limits the potential generality of the results owing to known and unknown biases in the data. This problem is commonplace in landscape ecology studies, and is not easily solved, notwithstanding randomized, landscape-scale sampling methods that have been described and advocated (e.g., Johnson and Gutsell 1994). These designs can be inappropriate for some objectives, or highly impractical, inefficient, or simply impossible to implement due to high variability of most mountainous landscapes, and the relative rarity of appropriate sampling sites and specimens needed for some types of studies. This is particularly true for paleoecological reconstructions, where a primary goal is to obtain the longest and most complete records of past environmental patterns and processes as can be found (Brown and Sieg 1996). Obtaining useful paleoecological records across landscapes calls for a "searching" strategy to locate the sites and specimens that will actually meet the data and sampling requirements. In practice, our strategy involved selecting representative sites within larger areas, as best we could subjectively identify them, and obtaining long and complete inventories of fire events by systematically searching for appropriate fire-scar specimens within sites.

Ultimately, our results from the selected sites constitute "case histories." We will demonstrate with examples how informative these case histories can be, both individually and in aggregate, about disturbance pattern and processes across a range of spatial scales from stands, to watersheds, to mountain ranges. The key to this strategy is the high temporal resolution and accuracy of dendrochronologically dated fire events (to the year or season) which facilitate inter-comparisons of numerous event inventories across space. Patterns of synchrony and asynchrony of such well-dated disturbance events across space reveal the extent of the disturbance process, the relative importance of internal versus external controls, and the spatial-scale of the disturbance effects. These cross-scale analyses allows us to extend and generalize our interpretations of disturbance regime patterns and changes (Baisan and Swetnam 1990, Grissino-Mayer 1995, Morino 1996, Swetnam 1993, Swetnam and Baisan 1996, Swetnam and Betancourt 1990, Touchan and others 1995, and other papers listed in Table 1).

Most study sites ranged in size from about 10 to 100 hectares. A few study sites encompassed collections from sets of fire-scarred trees in stands distributed over areas ranging from about 1,000 to 5,000 hectares (Rhyolite Canyon, Animas Mountain , and Mica Mountain sites). These were minimum sizes of areas that the fire histories probably represented, based upon polygons defined by the location of sampled fire-scarred trees. Many of the fires recorded within sites probably spread from distant locations, so the total area of the "fireshed" in which sites were located were much larger, but the size and boundaries of these firesheds cannot be accurately estimated with our current knowledge.

Collection and Analysis of Fire-Scar Specimens and Data

The seventeen fire interval distributions described here are based upon composite fire chronologies (Dieterich 1980) from numerous fire-scarred trees collected within each of the Madrean Borderlands study sites (Fig. 1, Table 1). We systematically searched for and collected fire-scarred trees with a maximum number of visible, well-preserved fire scars that were widely distributed throughout the study sites. Specimens were collected from dead trees (stumps, logs, and snags) and living trees in order to improve spatial coverage within sites, and to lengthen the temporal record of the fire histories (Baisan and Swetnam 1990). Partial and full cross sections were obtained using a chainsaw (Arno and Sneck 1977). The sections were finely sanded with belt sanders to see cell structure and fire scars within the rings The tree-ring widths were carefully crossdated using standard techniques (Stokes and Smiley 1968, Swetnam and others 1985). The fire scars were dated to the calendar year and identified to approximate season of occurrence based upon observations of the intra-ring position of the fire scars (Baisan and Swetnam 1990, Dieterich and Swetnam 1984,).

The fire-scar dates were entered in a specialized data base format and analyzed with a graphical and statistical analyses software package designed for this purpose (see Grissino-Mayer 1995; this software and manual are available via the Worldwide Web at http://www.ltrr.arizona.edu/). Statistical descrip-

tors of the fire interval distributions for a common period (1700-1900) were estimated for all sites. These descriptors included measures of central tendency (mean, median, Weibull median probability interval [WMPI]), range, and standard deviations. The WMPI, derived from a fitted Weibull model to the cumulative fire interval distribution (Grissino-Mayer 1995), is the interval (in years) at which there was approximately a 50 percent chance of a longer or shorter interval occurring during the summarized time period (1700-1900).

To evaluate fire interval patterns as a function of relative fire size we computed these statistics for interval distributions based upon fires recorded by:

1. All fires recorded, regardless of percentage of trees scarred; these were all fires occurring anywhere within the sampled area, regardless of inferred relative fire size;
2. Fires recorded by 10 percent or more of the fire-scarred trees, and a minimum of two trees; these were all fires recorded within the sites, except the smaller, and probably patchier fires recorded by only a single tree or a small percentage of trees (i.e., less than 10 percent);
3. Fires recorded by 25 percent or more of the sampled fire-scarred trees, and a minimum of two trees; these were generally the widespread fires that probably burned throughout most or all of the sampled area.

RESULTS

Statistical Descriptions of the Fire Interval Distributions

Summaries of fire interval statistics provide a generalized perspective of historical fire regimes in the Borderlands region (Table 2). These statistics suggest that some patterns were probably associated with changes in elevation, moisture, and forest type. Overall, however, the picture is one of considerable site-to-site variability. This variability of fire regimes among the same or similar forest types was most likely due to unique site differences in topographic setting and past land-use history (Swetnam and Baisan 1996). Some of these patterns will be discussed in the next section where we describe several case histories. In general, somewhat longer fire intervals occurred in higher elevation, relatively mesic, mixed-conifer forests (MC) than in lower elevation, relatively xeric, ponderosa pine forests. However, some MC and PIPO/MC stands sustained fire frequencies as high or higher than lower elevation PIPO stands (Table 2). The fire history study on Mount Graham in the Pinaleno Mountains is an example where a relatively high elevation mixed-conifer forest had a fire frequency similar to lower elevation pine forests (Grissino-Mayer and others 1995). We hypothesized that this was due to close proximity of these mixed-conifer stands to, dry, steep slopes, where fire could easily ignite and spread from many directions into the mixed-conifer zone.

Surface fires were quite common in nearly all montane forest types prior to about 1900. The maximum fire-free interval between fires, considering all fires within sites from 1700 to 1900 (see maximum fire intervals for "all" category in Table 2), was approximately 8 to 23 years. The exceptions were in Upper and Middle Rhyolite Canyon, where an unusually long interval occurred during the early 1800s, possibly due to a flood or debris flow event in these portions of the canyon that interrupted fuel continuity (Swetnam and others 1989, 1992). The minimum interval for the "all" fires category was one year in all sites. In other words, even in the smallest study sites, at least one, 1-year fire interval was detected during the period 1700-1900. Using the 25 percent category for fire dates (i.e., relatively widespread fires within sites) the minimum fire intervals ranged from 1 to 4 years, except in Rhyolite Canyon Middle (Table 2), where the minimum fire interval for fires recorded on 25 percent or more trees was 9 years. Interestingly, the smallest minimum fire interval for the 25 percent category, 1 year, occurred in the Rhyolite Canyon Lower and Rustler Park sites. We hypothesized that relatively high fire frequencies in these two Chiricahua Mountain sites, particularly in the late 1800s, may reflect increased fire occurrence due to Apache-set fires (Seklecki and others this volume, Swetnam and others 1989, 1992). Both areas were probable Apache camp sites or along often-used travel routes. Additional research is needed to test this hypothesis (e.g., by sampling in non-travel route areas, and by testing changes in climate-fire associations).

Several measures of central tendency are listed in Table 2 (mean, median, and Weibull median probability interval, or WMPI). Usually the measures of

Table 2. Summary descriptive statistics for Borderlands fire-scar chronologies, for the period 1700 to 1900 (except Sierra Ajos, 1700-1989).

Site name/ Mountain range	Fire size class	Mean	Median	WMPI	Min. fire interval	Max. fire interval	Stand dev.	Last widespread fire
Rhyolite Lower	all	6.17	6	5.41	1	15	3.80	
Chiricahua	10%	8.75	9	8.03	1	17	4.61	1886
	25%	9.21	10	8.77	1	17	4.35	
Pine Canyon	all	4.20	4	3.97	1	9	2.33	
Chiricahua	10%	5.10	4	4.79	1	11	2.80	1876
	25%	5.96	5	5.90	3	11	2.40	
Rhyolite Middle	all	8.30	7	6.78	1	33	7.28	
Chiricahua	10%	15.25	13	14.20	4	50	11.42	1886
	25%	17.90	14.5	17.08	9	50	11.86	
Sierra Ajos	all	4.26	4	4.01	1	18	2.66	
Ridge	10%	8.57	7	8.07	2	33	5.67	1972
Sierra Ajos	25%	9.60	8	9.12	2	33	6.00	
Sierra Ajos	all	4.04	3	3.79	1	22	3.05	
Saddle	10%	5.54	4	5.14	2	22	3.91	1972
Sierra Ajos	25%	5.88	5	5.47	2	22	4.05	
Rhyolite Upper	all	7.96	6.5	6.66	1	31	6.68	
Chiricahua	10%	12.64	12.5	12.22	4	31	6.68	1886
	25%	13.08	13	12.67	4	31	6.74	
Josephine Saddle	all	6.59	5	6.26	2	18	3.76	
Santa Rita	10%	8.24	7	7.94	3	21	4.24	1877
	25%	9.61	10	9.08	3	30	6.00	
Sawmill Canyon	all	4.88	4	4.67	2	13	2.62	
Huachuca	10%	5.93	5	5.57	2	22	3.97	1914
	25%	7.12	5	6.64	3	22	4.94	
Rose Canyon	all	5.50	5	5.26	1	15	2.92	
Santa Catalina	10%	7.33	6	7.01	2	16	3.80	1900
	25%	7.33	6	7.01	2	16	3.80	
Animas North	all	5.35	4	4.31	1	16	4.39	
Animas	10%	14.14	9	11.92	3	36	11.11	1879
	25%	16.50	12.5	14.65	4	41	11.77	
Animas South	all	7.42	6	6.61	1	21	5.02	
Animas	10%	14.33	14	12.66	2	32	9.27	1879
	25%	24.57	22	22.82	4	46	13.71	
Rustler Park	all	2.91	3	2.71	1	16	2.09	
Chiricahua	10%	3.85	3	3.56	1	16	2.51	1892
	25%	4.59	4	4.36	1	16	2.76	
Mica Mountain	all	2.95	3	2.67	1	9	1.94	
Rincon	10%	6.13	6	6.02	2	13	2.67	1893
	25%	7.32	7	7.13	2	13	3.29	
Pat Scott Peak	all	2.96	3	2.84	1	8	1.52	
Huachuca	10%	5.13	4	4.74	1	19	3.31	1899
	25%	9.75	7.5	8.76	3	29	3.41	
Lemmon Peak	all	6.60	5.5	6.03	1	17	4.08	
Santa Catalina	10%	8.61	9	7.94	2	17	4.80	1900
	25%	10.42	12	9.65	2	23	5.72	
Peter's Flat	all	6.10	4	5.24	1	22	4.69	
Pinaleno	10%	9.45	8.5	8.91	3	22	5.30	1893
	25%	12.60	12	12.35	3	22	5.25	
Camp Point	all	6.82	5	5.75	1	23	5.30	
Pinaleno	10%	8.52	8	7.73	2	23	5.60	1871
	25%	12.67	12	11.45	3	34	8.77	

central tendency were within one to a few years of each other. These measures were useful for generalizing typical fire intervals, however, the higher moments of the fire interval distributions (i.e., variance, range, skewness, kurtosis, etc.) may be of greater ecological importance than the central tendency descriptors. Moreover, distributions with similar means (medians, WMPIs, etc.) can have very different shapes (Fig. 2). In terms of plant responses to different or changing fire regimes, the relatively rare occurrence of long intervals between fires (the right tail of distributions in Fig. 2) may have been particularly important in determining the successful recruitment and survivorship of individuals or cohorts. The essential point here is that it is unwise to focus solely on measures of central tendency over blocks of time (e.g., fire frequency or mean fire intervals) in comparing historical fire regimes, or in attempting to interpret the importance of past fire to communities.

The distribution of seasonal timing of past fires, as inferred from the intra-ring position of fire scars shows that most fires in the Borderlands occurred in the arid fore-summer (Fig. 3). Of a total 3,701 fire scars that were examined, 2,656, or 72%, were successfully classified according to one of the five intra-ring position classes. The remaining 28% could not be classified because of very small rings, decay, etc. The percentages shown (Fig. 3) are for the classified scars only (i.e., the classes sum to 100%). Approximate seasonal timing shown above the classes is based upon our knowledge of cambial phenology of trees in southern Arizona. This knowledge is improving as we continuously gather data from a set of trees monitored with dendrometers and dendrographs in the Chiricahua Mountains (Baisan and Swetnam 1994). At this time, our best estimates of the timing of past fires, based on a splitting of the intra-ring position into five categories, are within windows of about two weeks to one month. These approximate seasonal dates overlap for each succeeding intra-ring position, due to variability in the cambial growth onset dates, rates, and cessation dates in different years, sites, and species. In general, most fires occurred sometime between late April and late June (Fig. 3). This corresponds approximately to the arid foresummer and 20th century lightning fire season (Barrows 1978). There were, however, differences in the seasonal timing of fires during specific years (e.g., Baisan and Swetnam 1990, Swetnam and others 1989), for different time periods (Grissino-Mayer 1995, Grissino-Mayer and Swetnam 1995), and in different sites (Seklecki and others this volume), all of which provide clues about the nature of these fires and fire regimes.

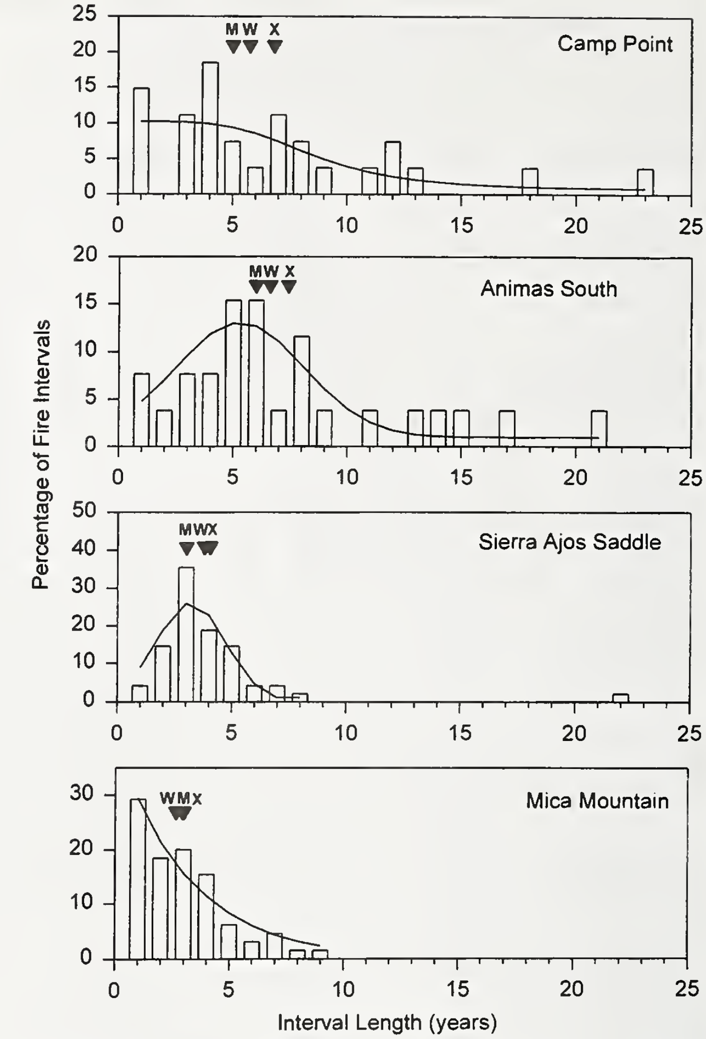

Figure 2. Examples of fire interval distributions from four study sites. The arrowheads show three difference measures of central tendency: M = median, W = Weibull median probability interval, and X = mean.

The ecological implications and importance of different fire interval distributions and fire seasons are not well understood. There probably is some predictability to the combination of plant species, age, size, and arrangement found in areas with particular types of historical fire distributions. However, there are no completed studies coupling data on such ecological *patterns* and *process* in the Southwestern U.S. over periods of centuries. (But see Danzer and others this volume, and Villanueva-Diaz and McPherson 1995, and this volume, for on-going stud-

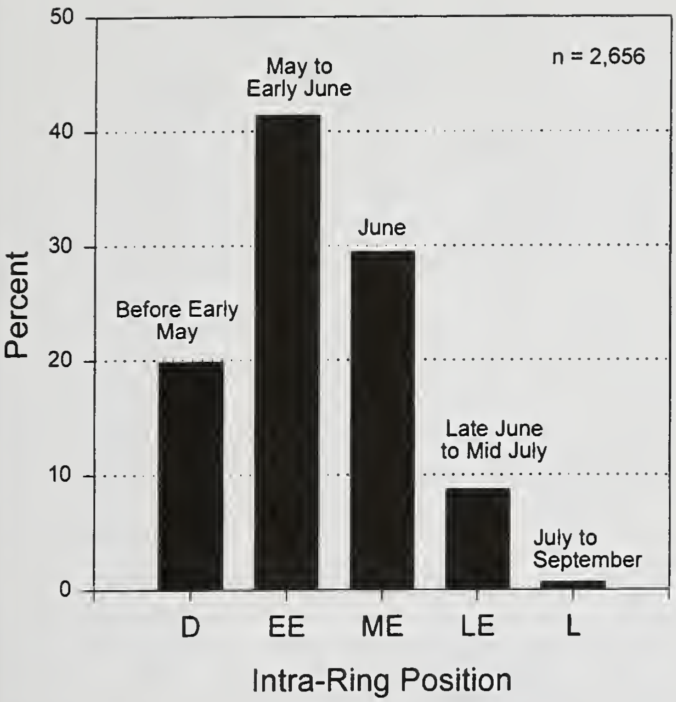

Figure 3. The distribution of intra-ring fire-scar position for the seventeen study sites in the Borderlands. The intra-ring classes are: D = dormant season scar; on ring boundary, EE = early-earlywood scar; within first 1/3 of earlywood, ME = middle-earlywood scar, within second 1/3 of the earlywood, LE = late-earlywood; within third 1/3 of the earlywood, L = latewood; within the latewood. (See text for explanation of seasonal interpretations.)

ies. Also see Barton [1993, 1994] for a very insightful set of observations and experiments relating to fire, micro-climate, and tree biogeography in the Chiricahua Mountains.)

Statistical summaries of fire interval or seasonal distributions (process histories) are useful for comparative purposes (e.g. Table 2, Fig. 2, Fig. 3) and ultimately these data and comparisons will help us develop and test predictive models of fire regime-ecosystem dynamics. For example, Miller (1995) has used fire interval and relative size data from fire history studies in the Sierra Nevada to test a process-based model of forest dynamics (ZELIG). A similar analysis is underway for mixed-conifer forests in the Sacramento Mountains of New Mexico (C. Regan, pers. comm.).

Summaries or generalizations of the fire interval and seasonal distributions over time are necessary first steps in our search for fire regime and ecosystem patterns. There is, however, another aspect of fire history that is at least equally important. This aspect is the fundamentally historical and chronological nature of fire event data. In the following sections we further document and describe Borderlands fire regimes with specific fire chronology examples. These three case histories illustrate both general patterns observed in other Borderlands sites, as well as their own unique, narrative histories that demonstrate the explanatory power of chronology and contingency. The general patterns shown by each case history are indicated by the headings of the following subsections.

Frequent, Low Intensity Surface Fires, Interrupted Circa 1870-1900

Fire-scar chronologies from two forest stands in the Santa Catalina Mountains exemplify a typical pattern of abrupt cessation of widespread, low intensity, surface fires at about the time intensive livestock grazing began in the Borderlands (Fig. 4). This pattern is most clearly seen in the Palisades/Rose Canyon chronology (lower graph), where the last widespread fire occurred in 1900. Shorter fire intervals are also evident in this chronology from a relatively dry pine forest as compared with the higher elevation, wetter, mixed-conifer site on Mt. Lemmon (upper graph).

The last dates of widespread surface fires consistently recorded by fire scars in the Southwestern U.S. (Arizona and New Mexico) were from the late 1870s to early 1900s (Swetnam 1990, Swetnam and Baisan 1996, Table 2). Comparison of these dates in different mountain ranges with land-use histories indicates that at least two contingent historical factors were associated with the decline of surface fire regimes: (1) cessation of hostilities with Native Americans (e.g., Apaches in the Borderlands), and (2) the rise of the livestock industry. The second factor was at least in part dependent on the first, but it was also a function of larger-scale economic forces, particularly access to emerging markets and railroads (Wagoner 1961).

The negative effects of livestock grazing on surface fire regimes was due to the removal of fine fuels (grasses and forbs) essential for carrying frequent surface fires (1 or more fires per decade). Total removal of grasses was probably not necessary in many arid and semi-arid landscapes; surface fires may no longer have effectively spread across landscapes where already sparse surface fuels were fur-

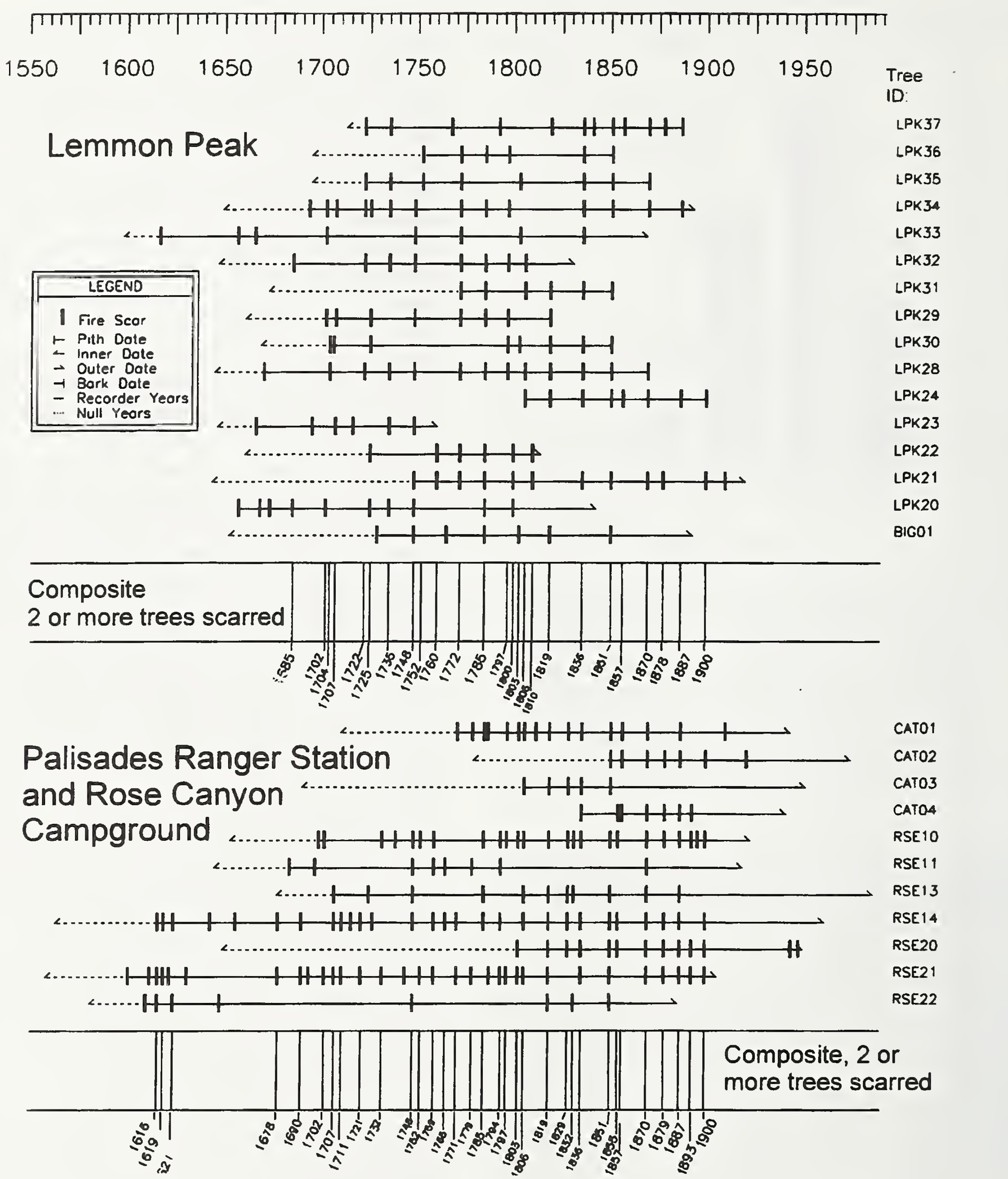

Figure 4. Master fire chronology charts for two sites in the Santa Catalina Mountains, Arizona. The horizontal lines are fire-scar records from individual trees and the vertical tick marks are fire-scar dates recorded on those trees. The composite graphs (long vertical lines at the base of each chart) show fire dates recorded by two or more fire-scarred trees. These chronologies are quite typical of pine and mixed-conifer fire chronologies in the Southwest, with widespread fire events within the sites indicated by synchronous fire dates among sampled trees, and a near cessation of such fires around the turn of the century.

ther reduced by grazing. New trails, fences, and roads associated with intensive grazing practices also disrupted fuel continuity, and hence the ability of fires to spread over large areas. Since site productivity is highly variable, and topography is also very important for fire spread, we would expect that different grazing intensities would have had variable effects on fire regime changes in different landscape situations.

The decline of widespread surface fire regimes in the Southwest was coincident with the boom in the livestock industry around the turn of the century (Bahre 1991, Denevan 1967, Wagoner 1952, 1961, Wilson 1995). We do not have good documentation of land-use histories for all of the different Borderlands mountain ranges, but there does seem to be some consistency in the historical comparisons where we have information. For example, because of the ruggedness of the mountain ranges, and the Apache threat, upper elevation forests of the Santa Catalina and Rincon mountains were probably not used for summer pastures until after about 1886 (Baisan 1990), when Geronimo surrendered to General Crook. The last widespread surface fires in study sites within these ranges were in 1900 and 1893, respectively (Table 1). In comparison, the Huachuca and Santa Rita Mountains (and especially the grasslands and woodlands at their bases) were heavy exploited for livestock grazing beginning in the 1870s because of the establishment of Fort Huachuca at the base of the Huachucas, and large ranching operations (Empire and Cienega Ranches) adjacent to the Santa Ritas (Wagoner 1961, Wilson 1995). The last widespread fire occurred in 1877 in the Josephine Saddle area of the Santa Ritas, and although a few widespread fires occurred after the late 1870s in the Huachuca sites, fire frequency clearly decreased ca. 1880 (Danzer and others, this volume).

This decline in widespread surface fires occurred even within areas where hostilities with Native Americans had already ceased some decades before, but widespread fires had continued to occur (e.g., parts of northern New Mexico). This argues that intensive livestock grazing was the primary and most common factor causing the end of frequent surface fire regimes, rather than lack of fire ignitions by Native Americans. The decadal-scale differences in timing of the last widespread fire in the different mountain ranges of the Southwest are even more convincing on this point. For example, widespread fire cessation dates in the early to mid-1800s, or earlier, have been documented in several areas of northern Arizona and New Mexico where herds of sheep, goats, cows, and horses were introduced by Spanish colonists, and later the Navajos (e.g., Baisan and Swetnam in prep., Savage and Swetnam 1990, Touchan and others 1995).

Fire suppression by government agencies did not begin in most places until after ca. 1910-1915, and arguably would not have been important in ending surface fire regimes without the collateral effects of livestock grazing (Leopold 1924). More effective fire suppression began after the 1930s when manpower increased (e.g., the Civilian Conservation Corps during the Great Depression) and many trails, guard stations, and lookout towers were built. Fire detection and fire fighting effectiveness increased again after World War II when surplus aircraft became available (Pyne 1982).

Climate change is an unlikely explanation for the abrupt surface fire cessation observed in Southwestern forests. If climate change was important in causing the extreme fire regime changes observed in so many locations we would expect regional climatic records (e.g., rainfall or temperature) to also show a major shift at this time. In general, they do not. A severe drought gripped the Southwest for a few years in the early 1890s and an extreme wet period occurred during the late 1910s and early 1920s (Fritts 1991, Sellers and others 1987), but fire regime changes in most sites were not consistently or closely synchronous with either of these two climatic events. In contrast, as will be discussed below, the ending dates of widespread surface fires corresponded more consistently and closely with the rise of intensive livestock grazing in each area. Moreover, surface fire regimes continued unaltered in mountains on the Mexican side of the border, and more-or-less unaltered in some remote areas on the U.S. side where intensive livestock grazing and/or effective fire suppression never occurred (e.g., isolated forest stands on "kipukas" in lava fields [Grissino-Mayer 1995]), but where similar regional climatic patterns prevailed.

Frequent, Low Intensity Surface Fires, Continuous Through the 20th Century

Fire-scar specimens have been collected in only a couple of mountain areas in northern Mexico. Re-

search in the Sierra de los Ajos (Dieterich 1983, Baisan and Swetnam 1995) shows that surface fires have occurred on this mountain for many centuries (since at least the mid 1400s). The most interesting and important feature of this history is the continued occurrence of frequent surface fires throughout the 20th century (Fig. 5). Fule and Covington (1995) report a similar continuation of widespread surface fires in pine forests in the state of Durango, although they note a shift to somewhat lower fire frequencies after ca. 1950. Other, recent fire-scar collections and observations in Mexican Borderlands mountain ranges (i.e., Sierra San Luis, Sierra del Tigre, Sierra Bocadehuachi, and in lower elevations of the Sierra de los Ajos) indicate that continuous, twentieth century surface fire regimes are not uncommon (Kaib, pers. obs.).

We sampled two sites in the Sierra de los Ajos. One was located in a single large forest stand on a saddle to the southwest of the highest peak in the mountains (Las Flores, 2620 m), and the other was in three adjacent forest stands along a ridge to the northeast of this peak. The two sites were about 4 kilometers apart. The saddle site had a somewhat higher fire frequency than the ridge site (Table 2, Fig. 5). We conjecture that this difference was related to landscape connectivity; the saddle site was subject to spreading fires from both sides of the main divide in this range, while the ridge site was subject to spreading fire from only one side.

The uninterrupted surface fire regimes in the Sierra de los Ajos have several important implications for our understanding of fire history in this region. First, continued, frequent surface fires suggest that

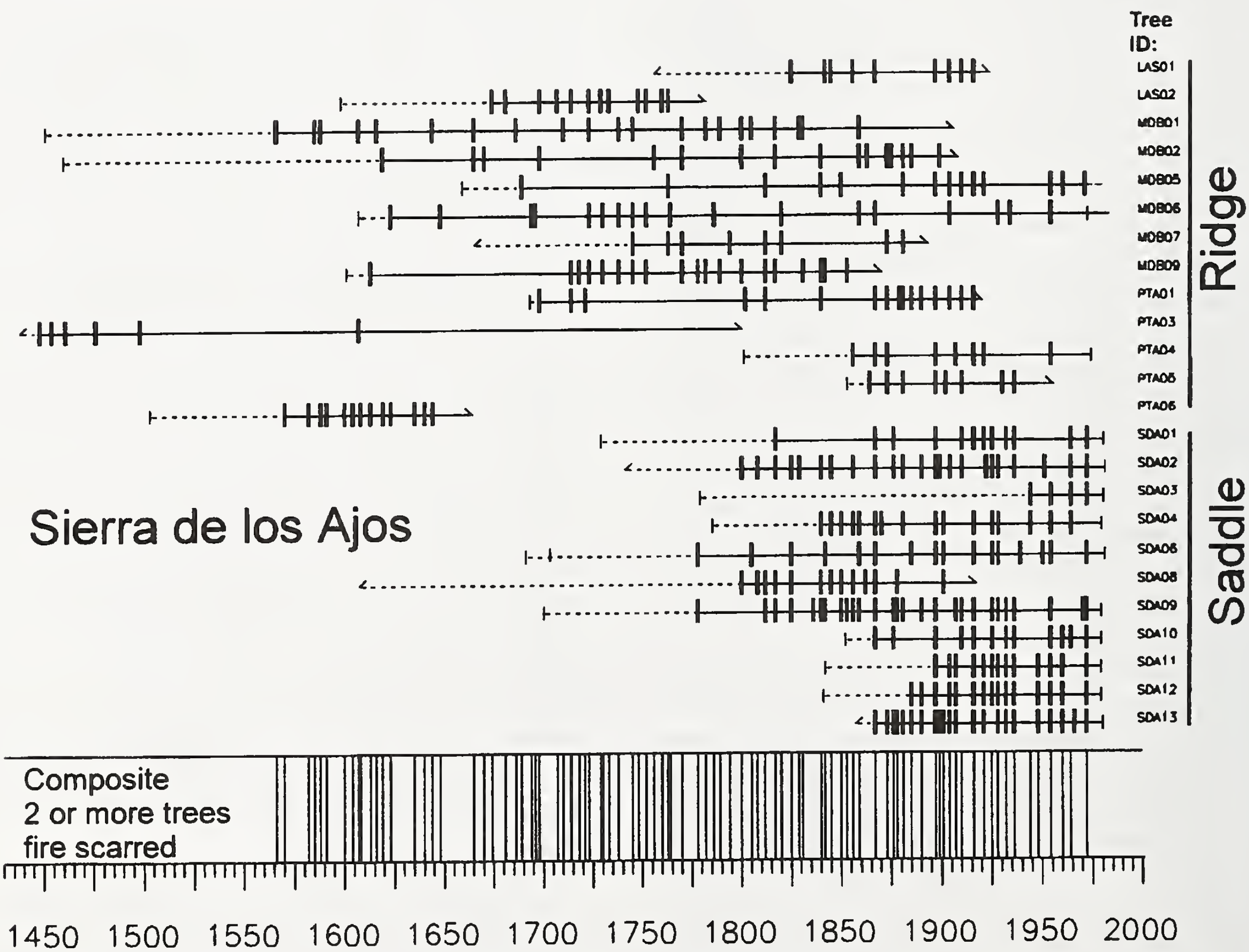

Figure 5. Master fire chronology charts for two sites in the Sierra de los Ajos, Sonora, Mexico. See caption and legend for Fig. 4. These chronologies document a continuous surface fire regime into the 20th century, a pattern observed in only a few isolated locations in the United States.

the final removal of the Apaches from the Borderlands in 1886 had no obvious effects on fire frequency in these mountains because human-set fires were not a primary, or limiting source of fire in these areas. Lightning fire ignitions were sufficiently frequent so that removal of Apache sources of fire had no distinguishable effect on fire frequency. Alternatively, Mexicans may have assumed the role of igniting fires on this range, as may have been previously carried out by Apaches. However, we doubt that either Mexicans, or Apaches before them, significantly influenced the fire regime. Today, these are remote mountains with no permanent settlements. The frequency of use of these mountains by Apaches is unknown. The area is currently grazed by a small number of cattle and there has been some timber harvesting, so it is possible that ranchers or loggers set some fires in the past. However, if either Apaches or Mexicans significantly influenced fire regimes in the past, and up to the present, we would expect greater variability in fire frequency through time than is evident (Fig. 5). Apachean movement and use of different Southwestern mountain ranges was probably sporadic, and partly dependent on changing political conditions (e.g., warfare). (See Morino [1995] for example, where temporal changes in fire frequency in the Organ Mountains, New Mexico were coincident with political change, and this was argued to be evidence of Apachean influence on past fire regimes.) If Mexicans assumed a hypothetical role of frequent landscape burning after the removal of Apaches they would have had to do this almost immediately after the Apaches were removed in the late 1800s, because no obvious shifts are observed at that time (Fig. 5). We would also expect that changes in climate affecting livestock production, politics (revolutions!), and economic fluctuations would have led to variable grazing activity through the 20th century, and the level of people's presence and burning activities within the mountain. The fire chronologies, however, do not show obvious changes that might be related to such human-related factors (Fig. 5). Finally, it is also evident from lightning detection records (e.g., Gosz and others 1995) that a sufficient number of lightning strikes occur in most areas of Southwestern and Borderlands mountain ranges to account for even the highest possible fire frequencies we can determine with fire scars (i.e., one fire per year).

A second finding derived from these fire chronologies is that grazing by cattle within these forests, at some level of intensity, seems to have had no obvious effect on the highly frequent, spreading surface fires. The important historical differences between the late 19th century effects of livestock grazing on fire regimes on the U.S. versus the Mexican side of the border were probably the numbers and types of animals that grazed these landscapes. We know very little about the grazing history or current number of animals in the Ajos, but we did observe small numbers of cattle grazing near and within our sampled sites when we collected our specimens in the early 1980s and again in the 1990s. We were quite impressed with the "open" character of the forests, the abundance of grasses, and evidence of recent surface fires. These conditions were reminiscent of Leopold's (1937) and Marshall's (1963) descriptions of northern Mexico forests when they visited there in the 1930s and 1950s. In contrast, some of the grasslands and savanna areas around the base of the Ajos appear to have been very overgrazed and eroded.

On the U.S. side of the border, there is abundant historical evidence that during the late 1800s livestock numbers were many times higher than today in almost all Southwestern mountain ranges and valleys, resulting in severe overgrazing. Southern Arizona was no exception in this regard (Bahre 1991, 1995, Leopold 1924, Wagoner 1961, Wilson 1995). In addition to cattle, large sheep and goat herds were also grazed in some areas of Southern Arizona (Bahre 1995, Hadley and others 1991, Wilson 1995). Open range, access to railheads, and a shift in ranching strategy from steer operations to cow-calf operations, may have been keys to the development of huge ranching operations on the U.S. side. The passage of mandatory range leasing laws in Texas in 1879 and 1883 created discontent and led to movement of ranchers and massive numbers of cattle to the unrestricted ranges of Arizona (Wagoner 1961). It is unlikely that these political and economic forces had much if any effect on Mexican ranges.

In summary, our explanation for the continued, frequent surface fires in the Sierra de los Ajos, and probably also for other Mexican Borderlands mountain ranges, was (1) the lack of intensive grazing by large numbers of cows, sheep, or goats in the higher elevations, and (2) a lack of effective fire suppression by the Mexican government. We doubt that the frequent fire regime recorded in our fire-scar history since the late 1800s was significantly influenced by

fires set by Mexican ranchers or loggers. The continued, frequent surface fires since 1886, when almost all Apaches were removed from the Borderlands, suggests that Apaches probably did not significantly increase fire frequencies in these areas above the fire frequencies that would have occurred anyway as a function of lightning ignitions and fuel dynamics.

Mixed Surface Fire and Crown Fires in Rugged Topography

The Animas Mountains fire chronologies provide an interesting contrast to the more typical frequent, widespread surface fire regimes reconstructed in most other Southwestern pine-dominant forests (Fig. 6). This complex history shows a pattern of "mixed" fire regimes, characterized by moderate frequency surface fires (about 3 to 15 year intervals) within individual forest stands, and widespread, higher intensity burns, including some stand-replacing fires, occurring at relatively lower frequency (about 20 to 50 year intervals) (Fig. 6).

The Animas Mountains fire-scar collection is one the largest from the Borderlands, both in terms of the numbers of trees sampled and in their broad spatial distribution. The master fire chronology (Fig. 6) is aggregated into clusters of 3 to 10 fire-scarred trees sampled within forest stands widely distributed over and around the highest peaks of the range (Fig. 7). This collection is comparable in extent to another mountain range-scale fire chronology from the Rincon Mountains (Baisan 1990, Baisan and Swetnam 1990). The Rincon Mountain fire chronology shows consistent, widespread fires (highly synchronous among trees and stands) at intervals of about 3 to 8 years. Little evidence for high intensity, long interval fires in the presettlement era was present in the Rincons. In contrast, the Animas chronology shows considerably less synchrony of surface fire dates among dispersed stands than the Rincon chronology, but other evidence points to occasional, mountain-wide (synchronized), high intensity burns.

The 1989 wildfire in the Animas range may have been an analog for the earlier synchronous moun-

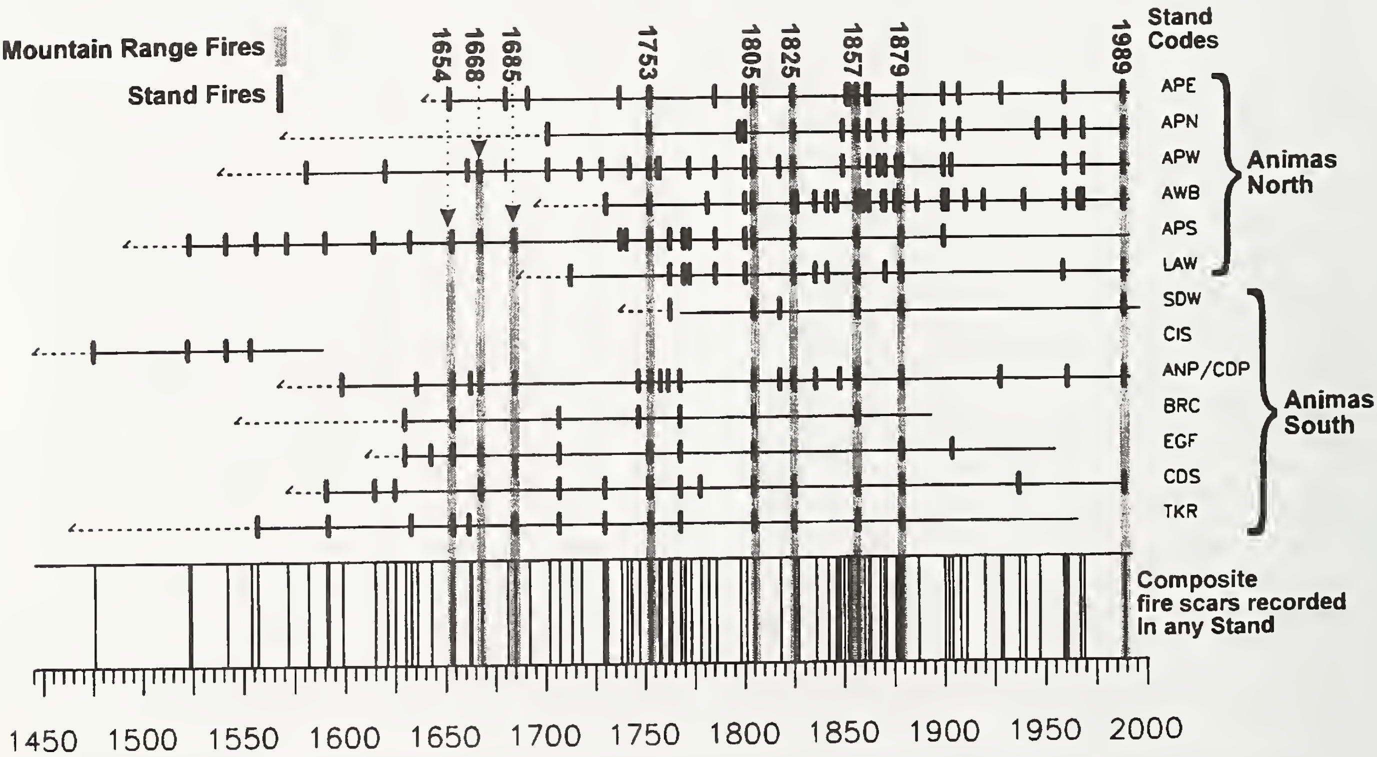

Figure 6. Master fire chronology chart for the Animas Mountains, New Mexico. See caption and legend for Fig. 4. The horizontal lines in this case are composite fire-scar records from groups of fire-scarred trees (three to 10 trees) from small stands distributed around the top of this range. The long vertical gray lines show years in which fires swept through most or all stands. See Fig. 7 for a map showing the spatial distributions of sampled stands.

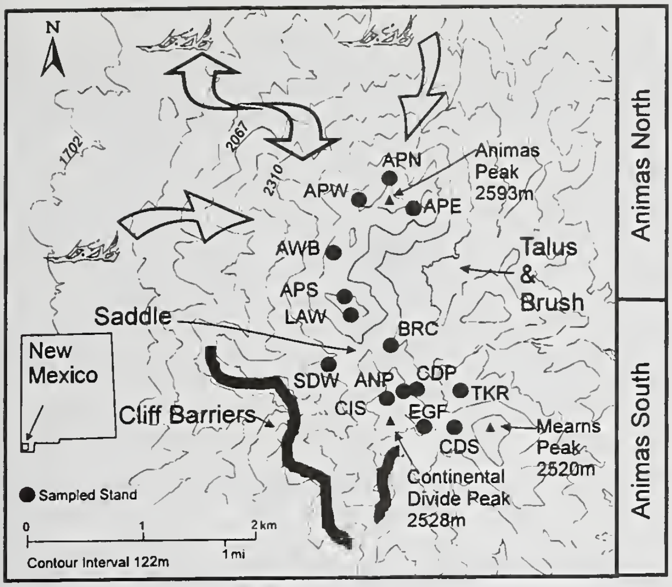

Figure 7. Topographic map of the Animas Mountains. Contour intervals are about 120 meters. Three letter codes show stand locations (corresponding to stand composites in Fig. 4). The northern stand group on Animas Peak was subject to spreading fires across the elevation gradient on the north side of the range. The southern stand group was more isolated from fires spreading from the grasslands to the west by cliffs.

tain-wide fires evident in the chronology (Fig. 6). The 1989 fire was ignited by lightning on June 15 in the foothills along the northern margin of the range. Suppression efforts did not begin until after the fire had spread over at least 10,000 hectares encompassing the whole array of plant communities from grasslands through mixed-conifer forests. Fire effects varied from light intensity surface burns with minimal impacts on overstory trees to total destruction of the forest canopy and understory vegetation. We do not have precise measurements of the sizes of the high intensity burn patches, but some appeared to be on the order of about 200 to 500 hectares in size. Our evidence for the 1989 analog conjecture is:

1. The similarity in the synchroneity of certain fire years (particularly 1753, 1805, 1825, 1857, and 1879, and 1989),
2. Dates of tree mortality events corresponding to some of these fires, and
3. Dates of tree recruitment following some of these fires (Baisan and Swetnam 1995).

Villanueva-Diaz and McPherson's data (1995, this volume) on tree age structure from several stands in the Animas also suggest that tree recruitment tended to follow large fires, particularly the 1879 burn. This is the pattern we would expect if stand structures were opened by relatively high intensity burns.

Other spatio-temporal patterns in the chronology point to the importance of landscape connectivity and land-use history. It is evident, for example, that surface fires were more frequent in some of the Animas Peak stands (upper 4 stands in Figs. 6 and 7) than in the stands located further to the south. This pattern was probably due to the continuous, unbroken topography and fuels leading from grasslands and woodlands on bajadas at the base of the mountains up to the summit of Animas Peak on this north side. Fires could spread (and did in 1989) unhindered by topography from any ignition point along this long elevation gradient. It is notable that our northernmost sampled site (APN), located on a north facing slope, is an almost pure ponderosa pine stand, with widely spaced trees and a grassy understory. In contrast, other stands south of Animas Peak (Fig. 7) are more isolated from spreading fires by natural fire barriers, such as cliffs along the west-facing escarpment of the range, and very steep, treeless, talus slopes in some areas along the east slope. Higher elevation north-facing stands in this southern area tend to be closed-canopy mixed-conifer. Other areas along the east and south sides, and in some cases located between our sampled stands, currently sustain only scattered oak-brush fields inter-mixed with talus slopes. Hence, southerly located stands in the range are more isolated from spreading fires than stands on the northern end of the range (Fig. 7).

Differences in 20th century fire occurrence patterns between the north and south stands reveals the importance of both land-use history and landscape/fuel connectivity. The first significant ranching in the Animas Valley and Range probably began in the late 1880s. The last widespread fire before 1989 in the southerly stands was in 1879. However, some fires continued to occur in the northern stands (Fig 6). Again, we ascribe this pattern to the continuity of topography and fuels on this side of the mountain. Despite the effects of an unknown, but probably low level of livestock grazing, these areas still had sufficient fuels to carry fire within and between open grassy forests on these slopes. Fires may have come from, or burned down into, the lower elevation woodlands and grasslands on this side. In comparison, fire regimes in the more isolated, southerly stands seem

to have been impacted immediately by the onset of livestock grazing in the 1880s, with a cessation of widespread fires, and the occurrence of only a few fires within stands during the 20th century (Fig. 6).

A spring in the relatively flat and grassy saddle between the north and south stands, and another spring just over the saddle on the east side, may have been the main impetus for moving livestock onto and through this part of the mountain seasonally. The livestock grazing (and trails, etc.) may have had such a pronounced effect here, but not in the northern stands, because of already sparse and disconnected fuels within and between the southerly stands. In other words, prior to the livestock grazing era there was probably sufficient fuel to sustain some spreading fires within and between the southern stands, albeit at a somewhat lower frequency than the northern stands (Fig. 6). After livestock grazing began, fuel amounts (grasses and forbs) and connectivity were reduced below a threshold necessary for sustaining surface fire regimes in these relatively isolated stands.

From the early 1910s to the 1950s the Animas range was under U. S. Forest Service jurisdiction. A primitive fire lookout was set up on one of the peaks in the early years but was probably manned only during some periods. The less frequent and synchronous fires between ca. 1910 to the 1950s in the Animas Peak stands may reflect effects of these minimal fire suppression efforts. The Animas range and surrounding lands were transferred to private hands in the late 1950s. After that time, limited fire suppression assistance came from state and federal agencies, but some fires probably reached large sizes on the northern side, and other areas within and around the mountain before much attention was paid to them in this remote corner of New Mexico.

Finally, we are left with one more piece to this complex puzzle of past fire regime patterns in the Animas Mountains. How is it that, during certain years, at relatively long intervals, fires seem to have swept over most or all of the stands, even though the southern stands were relatively isolated? Our explanation is that fuel connectivity (continuity and amount) between stands built up relatively slowly, so that at intervals of about 20 to 50 years conditions were primed for widespread, mountain-wide fires. Climatic conditions (daily to seasonal to interannual contingent events) conducive to fire ignition and spread promoted these mountain-wide events. A comparison with regional, tree-ring based drought reconstructions (Meko and others 1993) confirmed that the six mountain-wide fire events since 1700 all occurred during moderate to severe drought years. The 1879 and 1989 fire years were particularly notable as extreme drought and regional-scale fire years throughout the Southwest (Swetnam and Baisan 1996). The 110-year hiatus between the 1879 and 1989 fires was probably due to the combination of livestock grazing on the mountain preventing or slowing the within and inter-stand fuel build up, and the limited fire suppression efforts.

DISCUSSION

While ecosystem patterns and processes have certainly changed, as they always have, we would ignore their history at our own peril. Indeed, dismissal of the past as "irrelevant to current situations" early in this century by some forest scientists and managers was partly to blame for land management policies (e.g., attempts to totally eradicate fire) that have led directly or indirectly to many of the severe fire, insect, and pathogen problems we face today. Similarly, we feel it is a mistake to disregard the value of historical-ecological perspectives because they are potentially complicated by multiple interacting factors, or confounded by issues of scaling, or past human influences (e.g., Native Americans). Change is indeed a fundamental property of ecosystems — even without human intervention (Sprugel 1991). It is also true that the frequency and magnitude of past ecological change on any given landscape is dependent on the scale of our perspectives. Generally, longer temporal and larger spatial scale perspectives encompass changes of greater magnitude. These scaling complications, however, do not prevent us from using historical-ecological data to identify unsustainable recent or past changes, or for recognizing the causes and consequences of such changes. In fact, such identification and recognition are often impossible without historical-ecological data and perspectives.

The 20th century shift to increasingly large and intense crown fires in Southwestern ponderosa pine forests is a case in point (Covington and Moore 1994). This pattern is almost certainly due to fire suppression and subsequent accumulation of live and dead fuels. The historically and ecologically anomalous nature of these changes are indicated by documen-

tary and ecological studies (e.g., Cooper 1960, Weaver 1951), and by numerous fire history studies in Southwestern ponderosa pine forests (Swetnam and Baisan 1996). These patterns are especially evident as presettlement histories are contrasted with stand-replacement fire regimes occurring with increasing frequency in the 20th century in the same forests.

The main point of this example is that historical reconstructions and perspectives provide hard evidence for the reality of current extreme, unsustainable changes, as well as powerful explanations for their probable historical roots. These perspectives and explanations can provide direct scientific support to management decisions aimed toward restoring ecosystem processes and structures to more desirable and ecologically appropriate states.

Although we have learned something about high intensity fires, we do not yet have a clear understanding of the long-term role of these types of fire regimes in Borderlands mountain ranges, particularly the sizes of vegetation patches burned by such fires. We suspect that the 1989-type burn that occurred in the Animas may not have been historically or ecologically anomalous in this range, but we have doubts that this is true for the 1994 Rattlesnake burn in the Chiricahua Mountains or some other high intensity crown fires in southern Arizona ranges in the past decade. This is based on two observations. First, we lack evidence for fires burning at the intensities and patch sizes of the Rattlesnake and other fires within these ranges in the past (i.e., before ca. 1910). Such evidence would be recovering vegetation patches of these sizes (thousands of hectares), such as aspen stands, or conifer forests in varying seral, successional states, with at least fragmentary remnants (charred, snags, logs, or stumps) of the old, burned forest still present. Granted, as the old saying goes, "absence of evidence is not evidence of absence." However, within the limits of the preservation of fire-killed tree boles, and the time it would take for overstory trees to grow up into a structure that would hide the signs of a previous stand-replacing fire (perhaps 200 to 300 years or longer), we can be reasonably confident that canopy gaps the size of some created by the Rattlesnake fire, were rare to non-existent in the past few centuries. Second, we have fire-scarred specimens from a few locations (i.e., in the Rincons and Chiricahuas) where, clearly, only low intensity surface fire regimes persisted for the past three to five centuries. These stands, including the fire-scarred trees we sampled, were totally incinerated by recent crown fires.

We believe these facts constitute a "wake-up" call to managers, scientists, and the public. Some may argue that such extreme, historical-ecological changes should not necessarily concern us. Is a brush field slowly succeeding back to a conifer forest after a crown fire inherently less valuable or desirable than the conifer forest that burned? Some also argue that we should now just step back, and let nature take its course. However, heat damage to soils (e.g., loss of nutrients) probably occurred during the recent high intensity fires, and considerable soil erosion is occurring in some locations. This may cause threshold changes that will prevent these ecosystems from returning to forest or woodlands for many centuries, or millennia. The Earth has witnessed all manner of ecosystem changes, but heretofore, changes of these magnitudes and extent had their origin in non-human forces (e.g., climate change). Now, we humans may be the chief causes of these changes. This situation puts the responsibility squarely on our collective shoulders as land stewards to (1) learn whether these changes are truly, historically or ecologically anomalous, unsustainable, or undesirable (and these are not mutually exclusive conditions), and (2) to restore or maintain processes and structures in these biotic communities that will preserve their ecological legacy and integrity for future generations.

SUMMARY

1. Before 1900, surface fires occurred frequently (at least one fire per decade) in nearly all Borderland woodlands and forests with a pine component, but fire frequencies, and sizes were highly variable in both space and time. Mean fire intervals and other measures of central tendency and higher moments (variance, skewness, etc.) of pre-1900 fire interval distributions show some patterns that are, in part, functions of vegetation, elevation, and moisture relations. For example, higher elevation, relatively mesic, mixed-conifer forests tended to have longer intervals between fires than lower elevation, relatively xeric, pine dominant forests (Swetnam and Baisan 1996). High variability in fire history between sites was partly due to unique historical patterns (contingencies), such as time periods with unusually long or short fire intervals,

possibly due to natural events (e.g., floods or debris flows in middle and upper Rhyolite Canyon), or human-caused patterns (e.g., Apache augmented fire occurrence in lower Rhyolite and Rustler Park). Fire regimes are a result of continuous, repeatable processes that are at least partially generalizable and predictable, and unique, contingent events that are not strictly generalizable, or predictable.

2. Fire regimes in most Borderlands mountain ranges on the U.S. side changed drastically around the turn of the century. Frequent, widespread surface fires in most pine and mixed-conifer forests effectively ceased to occur between ca. 1870 and 1900. This change was initially caused by intensive livestock grazing, and subsequently, a combination of livestock grazing and fire suppression efforts by government agencies. Climatic change was probably not a primary factor in the initial cessation of widespread surface fires around the turn of the century, although the possibility remains that it contributed to the continued absence of fires in the late 1910s and 20s, when conditions were generally wet, and effective fire suppression infrastructure was not yet in place.

3. Frequent, widespread surface fire regimes persisted in the twentieth century in Borderlands mountain ranges on the Mexican side. These continued surface fire regimes were probably due to a lack of high intensity livestock grazing in pine forests, and a lack of effective fire suppression by Mexican government agencies. Persistent fire regimes in areas currently grazed by cattle also indicate that, under some livestock production systems, surface fires can consistently ignite and spread over the landscape. The continuous surface fire regimes after 1886 also suggest that Apaches were not a necessary source of fire in these or other mountain ranges to maintain high fire frequencies. Apaches may have increased fire frequencies in certain places and times above what they would have been with lightning ignitions alone, but more site-specific research and evidence are needed to identify those places and times.

4. Rugged, dissected Borderlands mountain ranges, such as the Animas Mountains, can sustain mixed fire regimes composed of both frequent surface fires, and relatively long interval, patchy crown fires. The 1989 fire in the Animas burned as a high intensity crown fire in patches. These patches were embedded in a much larger matrix of lower intensity surface fire. We hypothesize that this pattern may also have been sustained before 1900 in the Animas and in other, similar mountain ranges in the Borderlands. One implication of this type of fire regime is that fires burning at high intensity (crown fires) in patches, of some unknown size, may not be historically anomalous in some Borderlands mountain ranges.

Research Needs

Ultimately our decisions as land stewards will hinge, at least partly, on a better understanding of the past, which can guide us in understanding the consequences of our future actions. There are many things we need to learn more about.

1. We need to know more about the biotic consequences of different fire regimes. Specifically, what are the densities, age structures, distributions, and dynamics of plants and animals that are both a consequence and cause of different fire regimes in the Borderlands?

2. More knowledge is needed of fire history and fire effects in the highest and lowest elevation biotic communities, such as spruce-fir, upper elevation mixed-conifer, riparian zones, woodlands, savannas, and grasslands.

3. We need a better understanding of the historical and ecological role of high intensity, crown fires in mixed-conifer and spruce-fir, particularly the size distribution of landscape patches created by past high intensity fires.

4. We need to develop long-range (interseasonal and interannual) fire hazard forecasting models. A basis for these models may be the relation between the El Nino-Southern Oscillation, climate, and fire in the Southwest (Swetnam and Betancourt 1990, 1992) and the importance of interannual, lagging climate-fuel-fire relations (Swetnam and Baisan 1996). Such forecasting tools would be extremely valuable in fire management planning for extreme fire seasons, and for anticipating appropriate seasons and years for increased use of prescribed fire.

5. We need to apply the powerful new tools of remote sensing, geographic information systems, and dynamic, mechanistic simulation models to

improve our methods for reconstructing and interpreting past landscape history, and for gaining knowledge and understanding of current landscapes, fuels, and fire regimes.

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Human Desires and Fears in Ecologically Rational Wildland Fire Management

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Abstract.—While natural areas are generally perceived as desirable havens by city dwellers, the potential danger of fire is not always fully appreciated. People may correctly perceive the risk, but are unwilling to compromise their version of natural and aesthetically pleasing surroundings because environmental fears and desires are based largely on emotions rather than logic. Computer "visualization" technology may be more effective than words in motivating realistic response to fire danger in the wildland-urban interface.

Wildland areas have long been desired as places for recreation, solitude, scenic beauty and an escape from a crowded and stressful urban world. With increased mobility and the explosion of electronic communications, more and more people are extending their stay in the wildlands, realizing the dream of a "home in the country." While natural areas are generally perceived as a very desirable haven by city-dwellers, natural environments are in a constant state of flux. Not all the forces and interactions that shape natural processes are well understood, nor are they always benevolent to humans. Fire in the Madrean Province (and many other wildland settings) is one of the potentially dangerous forces of nature that newcomers to the wildland-urban interface do not always fully appreciate.

Wildland-urban interface fires continue to be responsible for the loss of lives and property in the US and other countries. Fire incidents continue to occur with alarming predictability; only the specific locations for the next disaster are uncertain. Adding to the toll is the cost of fighting wildland-urban interface fires, both in dollars and the lost lives of firefighters. Homeowners may recover the lost value of their houses through insurance or federal assistance programs. However, there is no replacement for lost lives, and survivors often suffer emotional distress from the loss of personal property and cherished mementos and from the disruption of their lives. Furthermore, the damage to the natural environmental setting, often the primary impetus for living in the wildland interface, may not recover for very long periods of time.

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PROBLEM

Census data reveals that rural populations are growing at more than twice the rate of urban areas. Furthermore, areas that were once isolated and remote, are achieving sub-urban and even urban status as more people manifest their dream of having a home in the country. Thus, people and human developments are rapidly expanding into areas that are subject to significant threat from wildfires. The situation is one in which there are increasing numbers of people in settings where they are exposed to substantial risk of fire.

Fire is, of course, a perfectly natural force (Vogl 1971b). Entire plant communities have adapted to the presence of natural fires; indeed, for some their very existence is a product of, and dependent upon, regular fire occurrences. Chaparral and prairie ecosystems and southwestern ponderosa pine are prominent examples of systems that are dependent upon periodic fire to refresh and continue the plant community (Wright, 1974).

Fire is an integral part of many environmental settings found desirable by humans, but humans have often not appreciated the beneficial role of fire—focusing instead on the destructive and threat-

ening aspects of fire. Decades of well-intended fire-suppression have resulted in an unnatural amount of fuel accumulation in many natural systems (Vogl 1971a, Wright & Bailey 1982). Leaves, branches, logs and other "litter" that would normally be consumed in periodic natural ground-fires are left to accumulate. Brush and dense stands of smaller trees that would be thinned by natural ground fires have tended to replace more open stands of larger, higher canopied trees, (Leopold et al., 1963).

Efforts to "protect nature" often result in increased risk to people. Fire suppression is often most vehement in areas that border housing sites. Potentially beneficial fires are quickly suppressed to avoid any risk to property and to prevent smoke and charring that homeowners find objectionable. Paradoxically, this is a perfect strategy for progressively increasing the hazard as stands become more dense and fuels coalesce horizontally and vertically (Dodge 1972). People show little sign of recognizing this hazard. Despite the best efforts of fire-safety experts, combustible houses continue to be built within thick stands of fire-prone trees and brush, with no apparent appreciation of the threat involved.

While much effort and publicity have been given to "preventing fire", experts generally agree that the most effective method for managing fire hazards in the wildland/urban interface is to reduce the fuels that feed the fire (Wright & Bailey 1982), and to adopt community designs and building standards that reduce the flammability of structures (Ramsay & McArthur 1991). While these guidelines would no doubt be effective safety measures, compliance is not overwhelmingly obvious in wildland/urban interface developments.

Typical developments allow trees and brush to grow close to, over and under structures, and flammable construction materials continue to be popular (wood shingle roofing being the most notorious) (Ramsay & McArthur 1991, McCarthy 1991). This may in part be due to a failure on the part of residents to recognize the "signs" of fire hazard, leading them to underestimate the potential for damage from a catastrophic fire. Indeed, there is evidence that people in fire-prone areas generally indicate little awareness of the risks to their properties (e.g., Cortner et al., 1990, Gardner et al., 1987).

It is also possible that residents' risk perceptions are essentially accurate, and that the lack of fire prevention actions reflects reasoned trade-offs in favor of less effort or cost, or in favor of what are perceived as more aesthetic "natural" surroundings. Gardner et al., (1985) noted that while sampled homeowners recognized the wisdom of California law requiring 30 to 100 feet around the home clear of vegetation, many admitted failing to comply with the regulation.

Scientific understanding of the biological processes of fuel generation and accumulation, and of the physical processes of fire propagation through these fuels provides a sound basis for an effective fire hazard reduction program (Daniel & Ferguson, 1991). However, effective implementation of any program of fire hazard reduction in the wildland/urban interface must necessarily rely on the participation and cooperation of individual residents and property owners. This support depends upon residents' perceptions and appreciation of the risks, acceptance of some responsibility for managing the risk, and on their willingness to comply with particular hazard reduction policies. The problem may be seen to be a trade-off between residents' risk perceptions, "fears," and potentially competing aesthetic and other environmental "desires."

It is possible that many wildland-urban interface residents are not cognizant of the risks represented by overhanging trees, dense brush and other fuels close to their homes. This would imply that they have somehow managed to avoid the many brochures, radio, television other public information programs that strongly press the need for clearing fire-prone vegetation away from structures. Alternatively, people may correctly perceive the risk, but are simply unwilling to compromise what they see as natural and aesthetically pleasing surroundings. Among the most important reasons people cite for moving into wildlands are a desire to be close to nature, to experience natural scenic beauty, and to increase privacy, solitude, and peace in their lives. Often, rural areas are perceived as a haven, safe from the threats of pollution, noise, and urban violence. It is not surprising, then, that wildland/urban interface residents are reluctant to make substantial changes to the natural environmental setting in the interest of increased safety. This reluctance may be especially strong if, as some fire safety guidelines seem to suggest, it is necessary to clear all vegetation for a substantial perimeter around the home.

A typical complaint of land managers when residents in the wildland-urban interface fail to heed

their warnings of fire and other hazards is "People just don't understand." However, human desires and fears may have relatively little to do with "understanding", as environmental perception research has shown.

Perception of environmental hazards.—Technical assessments of fire hazard are based on factors such as fuel loading, distribution, condition, ignition probability, and a number of situational factors including wind speed and direction, and fire suppression capabilities. It is not known whether or how these technical factors affect untrained observers' (residents') perception of fire risk, but the literature shows that peoples' risk perceptions often are not consistent with such technical assessments (e.g., Slovic et al., 1982).

People's assessment of risk depends on the perceived magnitude of expected damage, and on psychological attributes such as familiarity, dread, and degree of personal control (Slovic et al., 1986, Covello 1983, Vlek and Stallen 1981, Slovic 1987). The other factor in risk assessment is the estimated probability of occurrence of the damaging event. Whatever the magnitude of expected damage from an event, there is no risk if there is zero probability of the event occurring. Based on these general principles, perceived risk from wildfire would depend upon the perceiver's estimates of the extent and nature of damage expected, as well as their familiarity with fire and fire behavior and whether the perceiver believed that the consequences of fire could be controlled (e.g., by evacuation to save lives, fire fighting to save property, or insurance to replace losses). These factors would combine multiplicatively with the probability of occurrence estimate to determine perceived risk.

One obvious difference between interface residents' perceptions of wildfire risk and that of the fire-safety expert is the estimated (assumed) probability of occurrence. Residents typically estimate that probability to be quite low ("that doesn't happen here"), while fire experts tend to place that probability at very near 1.0 (the only chance factor is "when").

Environmental aesthetics.—Many studies have addressed human perceptions of the natural scenic beauty and environmental preferences in fire-prone wildland environments. These studies have shown a very consistent pattern of environmental preference among diverse public groups, both within and between different countries in the world. Moreover, consistent relationships have been found between measured forest characteristics, such as numbers, sizes and species of trees, volumes of downed wood, and density of vegetative ground cover and quantitative indices of perceived scenic beauty or aesthetic preference (e.g., Brown and Daniel 1986, Buhyoff et al 1986, Daniel & Boster 1976, Ribe, 1990, Schroeder & Daniel 1981).

More direct studies of the effects of fire on forest aesthetics (Anderson et al 1982, Taylor and Daniel 1984, Taylor et al 1986) have shown that forest areas are perceived as significantly less attractive (scenically and as sites for recreation activities) immediately after low intensity (prescribed) fires. However, scenic and recreation values quickly return to, and even exceed pre-fire values. For high intensity wildfires, on the other hand, very substantial and long-lasting degradation of scenic and recreational values is observed. While Taylor and Daniel (1984) showed that information regarding the ecological benefits of fire significantly increased public willingness to *tolerate* charring and other visible effects of fire in wildland areas, these same (informed) observers still perceived burned areas as less scenically and recreationally attractive. Thus, visitors to public wildland areas may be willing to accept short-term visual degradation by fire in return for ecological benefits and for longer term improvements in aesthetic values. It is not known, however, whether wildland/urban interface residents would be willing to make similar trade-offs, especially where their own properties are concerned.

Taken together, environmental perception research points to a potentially complementary relationship between aesthetic preferences and fire hazard reduction. Forest features associated with higher perceived scenic beauty are also known to be associated with reduced likelihood of catastrophic fires. For example, perceived scenic beauty tends to increase, and fire hazard decreases as forests tend toward a greater proportion of well-spaced large trees with little brush at eye level and lower volumes of downed dead wood. For fire to spread, there must be a continuous path of fuel for the fire to take. A discontinuity or break in the fuel, slows or stops the fire. Horizontal discontinuities can be man-made creations such as bare-earth firebreaks, or just open areas with little or no burnable fuels. Vertical discontinuities come about when the "fuel ladder" is broken, as when mechanical clearing or prescribed fire are used to remove

brush, litter and other components of the lowest rung of the ladder.

The spread of wildfires, then, can be impeded on either the vertical or the horizontal axis. Natural or built fire breaks prevent forward/outward movement, and breaking the fuel ladder prevents dangerous crown fires (Wright & Bailey, 1982). Fire managers often promote the use of horizontal fire breaks to protect rural homes, *viz* the prescribed vegetation-clear zone around the home. Vertical fire breaks are often ignored, but could be very effective. Removing the vertical ladder from grass to shrub to low drooping tree, may allow a fire to spread horizontally, but usually only as a non-destructive ground fire. Under these conditions, with modest effort to keep fuels from direct contact with the house, there is little danger of serious damage (Ramsay & McArthur 1991, McCarthy 1991).

The implied relationships between forest features that enhance perceived natural beauty and reduce fire hazard offer the promise of simultaneously achieving high levels of perceived aesthetic quality (desires) and substantial reductions in fire hazards (fears) in wildland-urban environments. This creates a possible win-win situation. Vegetation (fuel) patterns indicated for achieving high levels of perceived natural scenic beauty might not be as safe as a thirty meter bare-earth strip, but they would be considerably more consistent with interface residents' desires, and would be more likely to be supported and maintained.

APPROACH TO A SOLUTION

The general relationships observed between forest features that promote aesthetic values and that reduce fire hazard are clear enough, but how do we find the specific balance point between fears and desires in specific wildland/urban interface environments? Apparently, the most obvious answer to this question is to just ask them. Unfortunately, environmental fears and desires are based on emotions, not on logical conclusions. Words have proven a poor medium for eliciting realistic emotional responses, and words are also a poor medium for expression of important fears and desires (e.g., Buck, 1985, Zajonc, 1980).

To understand (and change) people's actions (and inactions) with regard to fire hazards, the processes by which these actions are determined must be taken into account. Often the strategy has been to "talk" to the residents (via brochures and other public information devices) in an attempt to increase their awareness of the technical factors and the magnitude of risks to their property, and to advise them about measures they can take to decrease this risk. The assumption apparently is that if residents can be given enough accurate information about fire hazards and how to reduce them, then they will behave more appropriately (i.e., like the experts suggest). This is somewhat analogous to a parent lecturing a child on the nutritional merits of spinach. Residents may perfectly well understand what they are told about fire hazards, in a technical sense, but when it comes to cutting down that overhanging tree (where Mr. Squirrel lives), or removing the brush (where the quail roost) on the downhill side of the house—they still won't eat their spinach!

Acknowledging the importance of the emotional components of wildland/urban interface residents' responses to fire hazards, how do we induce residents to manage their properties more appropriately? It might be possible to use a variety of techniques to raise residents' fears to a high enough level to motivate more effective compliance with fire safety prescriptions—some of the most enthusiastic brush clearing can be seen to occur as the fire approaches! The public official taking this approach, however, must be prepared to deal with potential political backlash effects, as Chicken Little learned long ago. Another approach is to accept the fact that both fears and desires must be accommodated, and to seek to help residents arrive at a more "optimum" trade-off between them. To do that, we must know how different landscape (fuel) management alternatives affect technical fire hazard/risk, and how those same alternatives affect residents' perceptions of aesthetic quality and of fire risk.

The environmental perception literature has documented that aesthetic perceptions can be effectively measured and predicted based on studies where public groups evaluate (via rating scales, choices or other responses) alternative forest areas represented by systematically sampled color photographs of the areas. This approach has been extended by the application of computer "visualization" technology. Alternative environmental conditions can be very precisely and realistically simulated, so that future (hypothetical) environments and environmental condi-

tions can be presented to public groups for their evaluation.

The same computer simulation technology can be used to represent alternative landscape/fuel conditions around and among wildland/urban interface developments. Thus, an array of different vegetation/fuel patterns, each associated with appropriate levels of technical fire risks as predicted by fire hazard/fire behavior models (e.g., speed of spread, intensity, flame height and other relevant fire parameters) could be simulated for a given site (e.g., the areas surrounding interface homesites). These visualizations could then be presented to groups of residents to determine their perception of the aesthetic quality of each "landscape plan," and (separately) to determine their perceptions of the fire risk associated with each "fuel pattern."

These perceptions of aesthetic values and fire risks could be used in a number of ways. First, by comparing perceived risk with the technical assessments provided by fire behavior models, specific consistencies and inconsistencies could be determined and used to better target public information and education efforts. Second, by analyzing the aesthetic and fire risk perceptions for the common set of wildland/urban interface conditions, it would be possible to find specific conditions that produce good ("optimum") trade-offs between desired conditions and conditions that reduce the risk of fire by technically known amounts. Finally, the visualizations themselves, along with their known technical fire risk assessments and their perceived aesthetic and risk indices, could be used to stimulate and guide a more meaningful, and more emotional, dialog between fire managers and residents. Such a dialog could greatly improve the chances of arriving at a more acceptable balance between residents' desires and fears, and of arriving at landscape/fuel management plans that managers, residents and the wildland/urban interface ecosystem can all support and maintain.

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Management of Fire, Laws and Regulations: Walking the Picket Fence

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Abstract.—Wildland fire managers walk a very difficult line in application of the science and art of fire management. With hundreds of laws, regulations, and policies to work within, the managers decisions are on the sharp point of criticism from persons desiring no management actions to those desiring to be totally protected from fire. Managers must develop the ability to work with ambiguity and a sense of personal "right" in their actions in order to survive in todays constantly changing world.

INTRODUCTION

To the on-the-ground manager of wildlands, it is difficult to appreciate the reasoning of persons that look at the land surrounding us and their belief that if we just "leave things alone" everything will be okay. This sense that we as managers have apparently messed up nature's plan over the past few years, and the only way to compensate is to back off and let nature return to control is just not acceptable. Knowing the consequences of allowing nature to take control again, managers must attempt to reinstate fire to a more natural role in our ecosystems, while balancing the socio-economic demands of our publics living near and within the wild lands we manage. This is a worldwide problem, as demonstrated by contributors from thirteen countries to the World Conference on Wildland Fires held in Aix-en-Provence, France, from 12 to 14 December, 1991 (Bourrinet, 1992).

THE REAL WORLD OF THE FIRE MANAGER

The Southwest area of the United States contains millions of acres of land containing vegetative ecosystems that have evolved over thousands of years in the presence of fire as a regularly occuring event. The Ponderosa Pine ecosystems of the Southwest are the most studied of our ecosystems from the standpoint of determining fire history. Dr. Tom Sweatnam of the University of Arizona Dendrochronoly Laboratory has studied our ecosystems extensively. He has also tied the fire histories recorded in tree rings with long term weather events. This has led to his work on the phenomenon of the El Nino and La Nina weather patterns that directly affect not only our weather, but our fire seasons in the Southwest.

The Past Leading to the Real World

The fire scar history that Dr. Sweatnam documents for our managers each time he provides assistance to us, tells us the same thing every time. Up until the time that European man became the dominant force in the Southwest, fires occured on thousands, probably millions of acres, every year throughout the Southwest. As European man entered the Southwest we began to graze the grasses and cut the timber. At the same time, our concept that fire was a destroyer of these resources and a threat to personal property began to influence the number of fires that burned over large acreage annually.

In the early part of the 20th century, as multiple large fires raged across the west and mid-west, burning millions of acres of timber and killing hundreds of people, there emerged a national sense that fire was an enemy that must be destroyed. As I look at the fire histories documented by Dr. Sweatnam and his associates, I find it interesting that at the time of the

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emergence of this national attitude that fire was an enemy, the influence of European man had essentially eliminated the impact of the naturally occuring "huge" fires in the Southwest. But the impact of the national attitude was applied throughout the nation in our laws and regulations relating to the management of federal lands during the decades from 1930 to 1960 (U.S.D.A. Forest Service, 1993).

There are many historical documents from across the Southwest which point out that at the turn of the century, millions of cattle and sheep were grazing the landscape. This impact affected almost every vegetative type in the Southwest, from the mixed conifer to the desert. This historical documentation of the quantum leap in use of grasses by cattle and sheep matches the abrupt end of the universal fire scar history that Dr. Sweatnam has found throughout the Southwest.

Growth of Laws Relating to the Real World of Today

The designation of public lands as forest reserves was first authorized by the Creative Act of 1891. This act authorized the President to set aside public lands for public purposes such as forest reserves. The Organic Administrative Act of 1897 expanded on this 1891 act by specifying the purposes for which forest reserves might be established and provided for their protection and management (U.S.D.A. Forest Service, 1993). The purposes for establishment of forest reserves were: to improve and protect the forest within their boundaries, securing favorable conditions of waterflows and providing a continuous supply of timber for the use and necessities of citizens of the United States.

From this humble beginning in the late 1800's, each succeeding decade saw an increase in laws relating to public lands. This trend of increases in legislation peaked in the 1970's, but there still exists a strong interest, and therefore a potential for new laws, relating to public land management.

Population Shifts Affect the Real World

The settlement of the Southwest occured during the final stages of the Agricultural Age in America. As an inducement to settle the west, miners could patent claims and mill sites and farmers could obtain title to lands by homesteading up to 160 acres. As the Industrial Age began to influence population shifts, many of these isolated homesteads and patented mining claims scattered throughout the Southwest were abandoned as a place of primary residence. The families moved to the city to become a part of this industrial period of prosperity. However, private property ownership and rights had been established, and as we will see later, would have profound impacts on the management of wildfire in the Southwest today.

As the Industrial Age has declined and transitioned into the Information Age, there has been another shift in populations in the Southwest. This shift has been the movement of families from the cities to small communities, as commuting long distances has become more acceptable. Many people want to raise their families now in a rural setting, with clean air, and clean water. A sort of back to our roots movement by those middle class workers that can afford the luxury of time and expense to commute to the city to work.

As the desire to move out of the city increased, the owners of the long unused homesteads and patented mining claims began to sub-divide and develop small bedroom communities surrounded by public lands. One of the primary characteristics of the homesteads in the Southwest was that a majority of these isolated parcels are located in valley bottoms, or meadows, surrounded by Ponderosa pine forests. The result is an appealing setting for the urbanite desiring to get back to nature and away from the urban sprawl.

As the small sub-divisions erupted with houses, it became quite evident to fire managers that we had a problem developing right here in "our" back yards. This problem grew throughout the United States during the last 30 years, but it was not until several destructive fires in California that the media picked up on the problem. The national press began to talk about the effects of fire on suburban populations in the late 70's as southern California made headlines with large scale chaparral fires roaring into sub-divisions, gobbling up houses by the hundreds. The concept of the wildland/urban interface became a part of our language. We now had a name for our "problem."

Houses, Fuels, and Other Assorted Problems in the Real World

To the average private citizen, a fire truck is a fire truck, is a fire truck. To the wildland fire manager,

and to the urban/suburban structural fire department manager, however, there are distinct differences in fire trucks. There are wildland fire engines, and there are structural fire engines; each must be equipped to fight fires that differ in fuels, hazards, terrain, and fire behavior. In addition to the engine equipment differences, there are other equipment and training differences. These other differences are so vast that they are actually liabilities when attempts are made to apply them in the wrong "world."

The turnout gear and breathing apparatus of the structral firefighter can cause extreme fatigue, heat buildup, and could lead to death if worn in a wildland fire setting. The Nomex fire resistant shirts and pants, and leather boots of the wildland firefighter provide no protection for the firefighter if attempts are made to wear inside a burning structure. The training necessary to attack fires in both of these distinctly different and complex "worlds" is also intrinsically different. The differences are so significant that only in very limited cases is it feasible or possible to cross train and outfit individual firefighters to perform both structural and wildland firefighting skills. To provide this duplication, in a safe, trained, operational mode, would require that there be two distinct, costly, mutually exclusive sets of fire gear. This is normally prohibitive to the average manager.

The increased fuel loadings, both standing and on the ground, in the forests associated with wildland/urban interface areas throughout the west has developed into an increasingly difficult fire suppression arena. With the addition of private houses this fire scene becomes not only more difficult from the standpoint of fire behavior, but also has a new hazard of multiple toxic/hazardous materials associated with residential fires. These are the sorts of hazards that require structural firefighters to wear breathing apparatus and turnout gear for protection.

The problem, of increased fuel loadings in the forest associated with wildland/urban interface areas, is further complicated by the addition of the structure itself-as a fuel loading. Consultation with the U.S. Forest Service Southwestern Regional Economist and Silviculturist indicate that the average United States private residence contains approximately 10,000 board feet of lumber. This translates, in a generality, to approximately 25,000 pounds of wood per home. With an average residential home population of 6-8 homes per acre, the fuel loading from the wood structure alone in the wildland/urban interface is approximately 85-90 tons per acre. When the interior furnishings are considered, the urban component of the wildland/urban interface can be considered to add approximately 100 tons of fuel per acre for the firefighter to contend with during suppression actions. The resulting rates of spreads and intensities associated with this type of "fuel" loading was graphically illustrated in the Oakland/Berkley Hills Fire of 1991. In one hour, 790 home were consumed. This translates to one home every 4.5 seconds (National Fire Protection Association, 1992).

A personal observation of the level of fire intensity that can result from the addition of structures into an already heavy fuel loading occured on the Tonto National Forest in the summer of 1990. The Dude Fire began just below the Mogollon Rim. This is a 1000 foot elevational change that cuts diagonally across Arizona. Normal summer fire weather, with associated southwest winds always drives fires like the Dude Fire up and over the rim. On this particular summer day, however, the normal winds, as well as everything else that we think of as normal, were not normal. Extreme temperatures, west winds, and fuel conditions that were explosively dry were the "norms" for this fire. As a result, the Dude Fire burned through an isolated subdivision containing approximately 60 homes. Fifty-four of these homes were evaporated by the fire. I say evaporated because in a normal structure fire, there is always some ash and other partially burned material left in the bounds of the former structure. In the case of the Dude Fire, there were several instances where the only remains of the structure was one or a few mostly consumed beams, some melted glass and metal, and nothing else. All the ash had been lifted by the energy of the fire column and carried into the atmosphere. This phemomena occurs only in the most intense fire situations, yet it occurred in a residential population. Television media coverage of urban fires provide clear evidence that the general public has little if any awareness of the overwhelming force of fires of this nature. As we interface with suburban forest residents, it becomes quite clear that they do not recognize nor understand the nature and degree of danger that the combination of structures and wildland fuels create.

The lack of understanding of the potential intensity, and danger of wildland/urban interface fires may explain why the NIMBY syndrome is alive and well in the urban interface. Everyone wants to have

their property protected, but the "Not In My Back Yard" syndrome is especially strong when wildland fire managers propose to underburn the forest next to a structure, or put smoke into private homes for more than a few minutes. There are few people that can tolerate smoke in their homes--even knowing that the smoke is necessary to keep the home itself from burning at a later date. The NIMBY syndrome is complicated and strengthened by the "It Will Never Happen Here" syndrome. These two syndromes taken together are extra sharp points along the picket fence that mangers must walk.

THE FIRE MANAGER'S DILEMMA

It is almost impossible to walk on top of a picket fence. It takes really good balance, as well as a tough sole on your shoes. There are only a few people in the world that are willing to even try. Luckily for the vast majority of private home owners around the world, those few that are willing to try to walk this picket fence are all located in thousands of local fire organizations.

This is the task of the wildland fire manager when considering the job of managing fire in the real world of today--To suffer the pain of walking the sharp points of a picket fence and maintaining balance while being "shot at from all sides."

This is My Neighborhood Too

Most wildland fire managers live in the wildland urban interface areas that they manage. Everything that they do in management or non-management affects them as well as their neighbors. The wildland fire manager is burdened by knowledge of fire behavior and effects as well as responsibility for management. The manager knows that the wildland urban interface area that is filled with fire dependant vegetation WILL BURN. Not IF--But WHEN is the question. This knowledge is always riding on the shoulders of the manager when dealing with the public--most of whom think that what they see is what has always been and will always be.

Every member of the neighborhood that the wildland fire manager lives in and works in has personal values. These values differ from person to person. Many of the values held by persons that live in the wildland urban interface are related to a desire to keep their back yards looking like it did when their dream home became a reality. Keeping this dream alive causes these individuals to act in ways that may not always appear rational to the wildland fire manager. The successful wildland fire manager must be able to empathize with these conflicting values, and become a communicator, a teacher, and a coach to these private land owners.

LEGAL ASPECTS OF MANAGING FIRE IN THE URBAN WILDLAND INTERFACE

The imaginary picket fence that a wildland fire manager must try to walk along can be described as being constructed of boards that represent the different values of the people occupying the interface area-on both sides of the line of public and private lands, the current climate of sue first, ask questions later, and the huge number of laws governing the management of public and private lands.

Just the Big Laws That Affect Us

A review of "The Principle Laws Relating to Forest Service Activities" (U.S. Department of Agriculture, 1993) shows that a staggering number of laws apply to wildland fire management. There are a total of 197 laws listed, and approximately 49 of these have possible application to management of wildland fire. From a practicing manager's perspective, however, we can avoid being overwhelmed by catagorizing the laws and recognizing and accepting their purposes. By this, I mean that we must recognize not only the effects of each law, but find ways to work within them. While this may seem to be an insurmountable challenge, your success in that endeaver will enhance both your capability and your career.

The primary impact law is the National Environmental Policy Act, P.L. 91-190 enacted January 1, 1970. This act guides our entire environmental evaluation process related to management of the land. If a wildland fire manager does not understand the importance of following the intent, as well as the specifics, of the NEPA process requirements, the potential for being knocked off the picket fence is almost certain. However, there is no reason for you to allow the NEPA process requirements to grind you down. By understanding the intent and goals in advance,

you can plan and manage your area with creativity and intelligence.

Another important and high impact law is the Endangered Species Act of 1973, P.L. 93-205, enacted December 28, 1973. When first enacted, managers knew that this law was going to have a high impact on management activities. But no one could envision that the impact would expand at the rate that has occured during the last decade. As demonstrated in recent events in the Southwest, this law, and judicial decrees relating to application of the law, have shut down multiple forest management activities for several months at a time. Interpretations of this law are likely to continue enlarging its effect on forest administration for several years.

An older law that did not have much of an impact on management until it was nearly 60 years old was the Preservation of American Antiquities Act of 1906, enacted June 8, 1906. We refer to this act as the Antiquities Act today. This law was strengthened and expanded in impact with the enactment on October, 15, 1966 of the National Historic Preservation Act of 1966. Fire managers have since learned to appreciate the role of the State Historic Preservation Officers (SHIPOs) in an approval role for our proposed management activities on federal lands.

The Clean Water Act of 1948, enacted June 30, 1948, and the Clean Air Act of 1955, enacted July 14, 1955, were also laws that sat on the books without significant impact on management activities until the late 1970's and early 1980's.

Another law that had little impact on management of federal activities for almost 20 years of existance is the Occupational Safety and Health Act of 1970, enacted December 29, 1970. Only in the last five years has the Occupational Safety and Health Administration begun to exert their authority to investigate accidents and safety practices in federal work places.

And last, but by no means least, the laws that protect us a federal employees in the performance of our duties. The original law, the Tort Claims Procedure ("Title 28, U.S.C.") enacted June 25, 1948 served to provide a comfort level for employees for 40 years. However, as the sue first ask questions later attitude began to grow in the United States, congress saw fit to enact the Federal Employees Liability Reform and Tort Compensation Act of 1988. Managers must still insure that any actions are within the scope of their employment, but as with most laws, there are few clear cut rules and many shades of gray.

Laws Are Double Edged Swords

Wildland fire managers are authorized to perform a variety of tasks. This authority lies in the laws that establish and govern our agencies. The cutting edge of this sword is this authority to act. The back side of the sword, which can also cut, is the responsibility to act within that authority. This responsibility to act within the law has been conceptually expanded by our publics today. The word of the wildland manager is no longer good enough that a specific activity is necessary. Our publics have increasingly exerted their authority as the "owners" of federal land to have a voice in what happens.

This desire to have an active part in the decision process is understandibly strong when the federal manager proposes a management activity in the "back yard" of one of these "owners." When this happens, not only is the "owner's" public land impacted, but often in the eyes of this "owner," private property, rights, and health and safety may be impacted. The values of these neighbors affect the decision process that takes place in public meetings, private interactions, in the news media presentations, and in the courts.

MANAGING TO STAY ON TOP OF THE PICKET FENCE

Wildland fire managers are inherently willing to be risk takers. But now more than ever, this characteristic must be balanced with the ability to recognize and address difficult issues and constrictions, and make sound decisions, for our agencies and our publics, affecting public land management.

Just as the laws regulating federal land management have increased, so, too, has the technology to predict impacts of management activities. We can now stand on site with a hand held calculator, and predict that under given fuel, site, and weather conditions, a given fire effect can be expected from application of prescribed fire. The science and technology of fire behavior prediction has improved our ability to design prescribed fire projects that are as safe as we can humanely predict, and has enabled us to make more well informed and intelligent decisions.

We as managers must gather appropriately trained and skilled applicators of this technology to use fire

as a management tool. We are legally bound to follow our plans with fully qualified personnel. Doing this, we are walking the picket fence of legal application of fire as a management tool.

There is a however to this picture. That however is that in spite of the best application of the science and technology of fire behavior and fire effects, there is another critical aspect of the job. That aspect remains the "art" of the application. This art is affected by all the intangible, unmeasurable, unseen but felt in the gut, things that occur on a minute to minute basis when we begin to walk the line with a drip torch in our hand and never stops until the last smoke is gone. This art is also affected by our public relations/ interpersonal communications skills that have been exercised in the preparation, and execution of a project.

Managers are always on the picket fence; trying to stay on top; always feeling the sharpness of the points of the fence trying to bore through the soles of our feet; and hoping that we know where the next picket really is and that when we put our foot down, we don't have a hole in the sole. At the same time, we must be aware of and work with a developing field of law and regulations, and dodge the slings and arrows of individual interpretations of laws, changes in predicted conditions, unpredicted weather conditions, and changes in personal attitudes and commitments that have occured since we commited ourselves to a project.

As the manager walks the picket fence, there is a growing awareness that the pickets are made up of our own personal values and the values of those we serve, as well as the legalities and scientific aspects of wildland management. The ability to stay upright is the art—and the whole world is an art critic. Just as a successful artist learns to create art with a sense of accomplishment that comes from within, the successful manager of wildland fire must become successful at maintaining a sense of personal success, or "right," that comes primarily from within. This internal locus of success or failure, tempered with a sense of responsibility to the owners of our public lands is the key to success both now and in the future.

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Forest Fires and Their Social and Economic Impacts

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Abstract.—This paper is a review of literature relating forest fire effects on humans and the environment. It presents some cases from traditional countries that have forest resources and their approach to fire management. and ends with a case in which a mathematical formula was employed to calculate forest fire damage in a tract of forest of 3.6 hectares in the state of Michoacan, Mexico.

INTRODUCTION

Fire has been a key element in the development of human beings. It has been used for protection against wild animals;for heating and cooking and for industrial production of goods. However, when fire is not controlled, when it becomes wild whether in cities or in areas of grazing, shrubs, forests, or jungles, it causes a tremendous impact. Effects on the environment are sometimes irreversible.

Fire kills living organisms growing in the forest, especially the vegetation and to a lesser extent animals. When trees are consumed, forest products are lost.

Forests provide additional services which are also lost with the occurrence of a fire, even pollution comes from it.

This paper presents a literature review about the social and economic effects of forest fires on international and domestic levels. It also presents an evaluation of a forest that was burned in 1993 in Ciudad Hidalgo, Michoacan, Mexico. The economic effects of the fire onthe forest were evaluated through the use of a mathematical model. Some of the mechanisms of the effects are also considered.

BACKGROUND

The international scenario of forest fires has been dramatic. Annually, countries report forest fires in the dry season and the multiple causes provoking them (FAO - UN, 1964):

- Smokers
- Intentional
- Electric discharge
- Slash and burn
- Military practices
- Researchers
- Fishermen
- Electric lines
- Charcoal kiln
- Forest utilization
- Recreation
- Land occupancy
- Lightning
- Other not recognized

In some countries, however, some factors are difficult to measure and difficult to prevent. In Israel, for example, people attribute the actual increase in fires to the following major factors:

1. Primary and secondary roads, forests, recreation areas are being expanded. As a result, forests are being visited with higher frequency, increasing the probability for forest fires.
2. Shepherds start fires to improve grasses and for reclamation of grazing areas.

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3. Many forest stands aong the edges of fertile valleys, where grain crops are grown. When farmers burn their residues, fire escapes frequently to forest areas.

4. Fires are set for policy reasons in the Middle East. Planting trees means to own the land. Policy adversaries set fires in an attempt to destroy this possession symbol.

Independent of the causes provoking forest fires, it is a fact that they appear. Mentioning the factors causing them is for the purpose of taking them into consideration in any evaluation, and to be concious that they are a latent danger.

In the period from 1980 to 1988, Europe suffered 42,100 fires annually that affected 585,000 hectares of mountains and forest areas. In North America, in the same period, it is reported 154,000 fires burned 3,478,200 hectares. According to Calabri (1991) the two continents averaged 200,000 fires and 4 million ha. devastated by fire each year.

In China, in May of 1987, a huge fire burned 1, 330,000 hectares in the northeastern part of the country, damaging 870,000 ha, 10,081 homes in three towns, and much equipment (Fuchs, 1988).

In Cuba, Oharriz and Valdez (1991) report the occurrence of 1,284 fires from 1981 to 1985. They were classified according to the cause that originated them: unknown 28%, smokers 24%, intentional 15%, and electric discharge 14%.

Diana (1991) presents statistics on forest fire occurrences and economic losses that are not complete. However, it is reported from 1983 to 1988, in the 6 million hectares planted of forests, fires burned about 201,263 hectares.

In Poland, Ryszard (1991) reported a distinct increase in forest fires. The average on a per year basis from 1981 to 1991 increased by 960 in comparison to the data of 1961-80. The average area burned yearly increased by 1,210 ha same time. In 1990 nearly 6,000 fires had burned about 9,000 ha of commercial forest.

Many efforts have been made in each country to prevent forest fires instead of combating them. There are also international agreements for research on forest fires, as well as fire suppression, planning and operation of fire management. Events like this symposium give opportunity to technicians, scientists, and society in general, to know the state of the art on forest fires. However, there is too much ahead to be done from the standpoint of operation and much more in research. People, especially in the developing countries, need to be educated about the importance and function of forest ecosystems, and how wild forest fires—or as we say in Mexico, uncontrolled forest fires—greatly diminish these functions.

The majority of research has been devoted to produce and apply models for forest fire prediction or for planning forest fire operations to reduce cost of suppression. Ryszard (1991) worked out, for example, a model based on laboratory and field studies in Poland for a fire spreading and suppression action. This model gives a mathematical relationship in a program of conversational type. It is also possible to apply the model in a computer. The most important parameters taken into consideration in the model are: wind velocity, fuel humidity, and quantity and duration of the fire, which allow the calculation of the speed of the fire front, its area, caloric value, height of the flames, quantity of heat emitted, and some other parameters. Application of the model gives the opportunity to calculate: the amount of extinguishing agent needed (water or foam) and number of fire suppression vehicles.

Gonzalez (1985) mentions that the level of expenditures, or justifiable investment level must be determined for fire management programs. Thus, he is convinced that it is wise to use simulation models for assessing the economic effectiveness of forest fire prevention programs. He mentions several simulation models that are being used in an economic evaluation, like the lowest cost plus losses, benefit cost, AIRPRO (simulation models determining productivity of aerial tankers), economic evaluation of AIRPRO, FOCUS (planning system for fire management), and NFMAS (domestic system analysis for fire management)

THE SCENARIO FOR MEXICO

Mexico has all the possible forest ecosystems, except the boreal. For many years, the Federal goverenment has reallocated people living in rural areas, forming new population centers, giving them land to own no matter whether this land had forest or jungle or other kind of natural resource. The first step for a peasant owning this land was to slash and burn in order to plant cash crops, such as corn, beans, sorghum and in some fortunate cases the establishment of orchards with classical tropical fruits. This

action provoked many huge forest fires burning much forest areas, including homes and some humans. The major cause of forest fires in the tropical forests is land use change, nowadays.

The Federal government has encouraged the attention toward forest disasters, specially on the wild forest fire programs, although they are not presently evaluated at all. However, this effort could make more and more people forest fire oriented.

From 1969 to 1994, there have been 152,073 forest fires, with an average of 6,083. The total are burned is 4,932,389 hectares with an annual average of 197,296 ha. The average affected surface by fire is 32 ha (SARH, 1994).

Major causes of forest fires have been carelessness and lack of deligence (agriculture and cattle - raising activities, bonfires, campfires, other productive activities near or in the forests, smokers, forest utilization activities, roadright), intentional, and other causes not identified.

Forest fires affect in one way or another, all forest ecosystems; however, they sometimes benefit natural resources. Table 1 depicts all types of damages and benefits in the three main forest ecosystems in Mexico.

Some of these damages are difficult to measure or evaluate quantitatively, but we are concious that they are there as soon as the fire occurs. Wee need to find a way to measure them, and research has to play an important role here.

According to SARH (1994), the last two national prevention campaigns in forest fire produced good results. To have an idea of the figures presented in the international context, let´s compare data for three countries, United States of America, Spain, and Mexico (table 2).

As we can see, there is a big difference in terms of affected area between the three countries as well as for the assigned resources. United States has a figure of 660,789 ha in 1994 with 380 million dollars. For Mexico, although the number of fires is almost twice that of USA, the affected surface in 1994 is one fifth of the corresponding figure of USA.

During 1994 in Mexico, the states reporting most forest fires are: Mexico (2061), Distrito Federal (1069), Michoacan (944), Chihuahua (626), Jalisco (531), Morelos (395), and Durango (302) (SARH, 1994).

The areas in hectares most affected were in the states of: Jalisco (19,377), Chiapas (16,673), Chihuahua (14,477), Zacatecas (12,091), Mexico (11,240), Nayarit (7,647) and Michoacan (5,584) (SARH, 1994).

We can say that the affected surface is not directly related to the number of fires; maybe some states have much more equipment to suppress fires or people are more forest fire oriented and the equipment is more modern.

Table 1. Effects of Forest Fires

Vegetative types	Damages	Benefits
Temperate Forests (Pines, oaks)	• Limit or damage natural regeneration. • Destroy commercial trees, causing economic losses. • Old trees are subject to the attack of pests and diseases. • No valuable species are presented. • Damage the soil layers, especially on the surface.	• Help in the opening of cones to release seeds. • Control some plagues. • Grass is renewed thanks to the fire, making it edible. for livestock. • Reduce fuel material, diminishing the risk for major fires.
Jungles (Tropical Forests)	• Alter biodiversity. • Destroy commercial species generating economic losses. • Damage the soil	• Improve the availability of nutrients. • Help in the regeneration of certain species.
Vegetation of Arid Zones	• Damage severely the vegetation. • Reduce job sources such as harvesting (Oregano, Lechuguilla, Jojoba, etc.).	• Through the fire, some specie are turned edible like opuntias for livestock and wildlife. • Enhance the reproduction of some species.

Source: SARH, 1994. Resultados del Programa Nacional de Prevencion y Combate de Incendios Forestales 1969–1994.

Table 2. Information on forest fires in USA, Spain and Mexico.

Country	Number of fires		Affected area		Area/Fire		Assigned resources	Human
	Annual mean	1994	Annual mean	1994	Annual mean	1994	1994	1994
USA	39,916	41,721	453,942	660,789	11	16	380	22
Spain	15,282	8,081	208,359	265,134	14	33	48	36
Mexico	7,153	7,830	213,010	141,502	30	18	4	2

Note: for USA and Spain the average is referred to the last 5 years; for Mexico it is for the last 6 years.

Source: SARH, 1994. Resultados del Programa Nacional de Prevencion y Combate de Incendios Forestales 1989–1994.

The Model And its Application

In the process of looking for a way to evaluate the consequences of a wild forest fire, we found a mathematical model used in Spain. We in Mexico had a fire in a commercial forest area in the state of Michoacan in 1993 which burned a 3.6 ha area. This area was being utilized or harvested because it was a young stand.

The formula we will be presenting calculates losses of forest products from a forest that is not under utilization i.e, not ready for cutting.

Damages in forest without commercial utilization are estimated equal to the necessary investment to establish a new forest, taking into consideration the capital soil, expenses in conservation, and the regenerating cost. It is considered also, to have a forest after the fire, i.e. the continuity of the forest resource. The formula is:

$$F_r = K\, S_r\, P_T\, V_T\, [\,((1+t)^{**}e_r\,) - 1)] \,/\, [\,(1+t)^{**}T]$$

Where:

F_r = Damage in a commercial forest without utilization or cut time.

K = Cover surface coefficient .

P = Average price per cubic meter, not used in this case.

S = Affected surface which contained trees without commercial utilization, i.e., not ready for cutting.

V = Wooden volume that is able to be produce in one hectare in the cicle cut.

e_r = Average age in years of the forest.

T = Turn of the forest.

t = Interest rate.

The total calculation gives us a figure of $ 75 per cubic meter. The calculation of the average volume in each hectare is of 62 m^3; so the total value for the 3.6 hectares is of $16,740.

This calculation is only one part of the possible damage that a fire could cause in a forest. However, the effects on the environment, wildlife, water, cover or soil, and beauty for recreation, are not considered in this model. Research must develop another one or several models considering the above.

FINAL REMARKS

Forest fires are an important phenomenon to be considered not only by forest managers but also by the entire society, whether local, regional, national, or international.

There is no strong metodology that can consider all the possible effects that forest fire causes, consequently research has to be performed to develop a strategy to measure the intangible effects that are up to now, not measurable.

Only when the above happens, will we have the opportunity to know the entire sum of effects a fire can cause, both qualitatively and quantitatively, and incorporate these as costs of production or cost for the entire society.

Meanwhile, we are going to underestimate the total relevance of forest fires, not giving them the importance they deserve.

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Effects of Fire on Montane Forest Ecosystems

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Abstract.—The effects of fire on the higher elevation Madrean ponderosa pine and mixed conifer ecosystems are examined. Morphological and physiological adaptations of ponderosa, Arizona, Apache, and Chihuahua pines are reviewed and related to the degree of fire tolerance by each species. Similar evaluations are made for mixed conifer species, including Douglas and white fir, Engelmann and blue spruce, southwestern white pine, and aspen. Restoration of fire to these ecosystems is difficult today because of the altered stand conditions resulting from decades of fire suppression. However, by understanding species' fire tolerances and ecological relationships, the manager will be able to implement efficient and effective burning programs to improve ecosystem health.

INTRODUCTION

The effects of fire on the montane ecosystems of the southwestern United States and northwestern Mexico are as varied as the biological diversity of the region. Consisting of mountains and valleys, aptly named "sky islands", the Madrean Archipelago lies as a series of stepping stones between two mountain chains, the Rocky Mountains and its plateaus to the north and the Sierra Madre mountains to the south (Warshall 1995). The biotic communities at the higher elevations support pine, mixed conifer and spruce-fir forests. The natural impacts of fires, insects, diseases, and climatic patterns have affected the establishment, growth, and stand characteristics of these forests (Gottfried et al. 1995). Changes in fire frequencies also have led to increased numbers of fires of greater intensity (Swetnam 1990). This paper will examine the role of fire on the higher elevation forests of this borderlands region and the implications for management of fire in the transition pine and the Canadian mixed conifer zones.

MADREAN MONTANE FOREST ECOSYSTEMS

A comprehensive listing of trees in the sky island region known to occur at elevations above the deserts and grasslands has been assembled by Felger and Johnson (1995). Their list, which defines trees as woody plants over 5 m (16.5 ft) in height, contains over 200 species representing 130 genera and 61 families. Many of the northern temperate tree species extend southward into the interior of Mexico at intermediate or higher elevations. Montane forests include both the transition pine and Canadian fir zones (Lowe 1961, 1964). Elevations range from about 2,000 m (6,560 ft) on the wetter and cooler sites to as high as 3,050 m (10,000 ft) on south slopes. The best and most characteristic development of montane forests in the Southwest occurs between 2,300 and 2,700 m (7,550 and 8,850 ft) (Pase and Brown 1982).

Mean annual precipitation can vary from 460 to 760 mm (18.1 to 30.0 in). Precipitation is bi-modal with moisture from the Gulf of Mexico causing summer monsoonal rains and Pacific frontal systems resulting in winter storms (Gottfried et al. 1995). Summer precipitation may contribute up to 70 percent of the annual total in southeastern Arizona (Bahre 1991). Although total annual precipitation decreases latitudinally south to north, it does increase with elevation due to orographic lifting. Soil moisture availability appears to be the key limiting factor for vegetative growth and species distribution in the region (Felger and Johnson 1995).

Mountain soils in the borderlands region are generally shallow and rocky with Precambrian and Tertiary geologic substratums. Isolated volcanic areas weather into finer textured, deeper soils (Hendricks 1985).

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The Madrean montane forest ecosystem can be conveniently divided into two major tree communities—the ponderosa pine transition forest, normally found at the lower elevations, and the mixed conifer forest of Douglas fir, white fir, and aspen at the cooler, higher elevations. The lower limits of the pine forest are typically in contact with pinyon-juniper woodlands, madrean evergreen woodlands, or grasslands (Pase and Brown 1982).

Ponderosa Pine Forest

Ponderosa pine is the most common montane forest tree in the Southwest. In southern Arizona, the three-needled Interior Ponderosa Pine (*Pinus ponderosa* var. *scopulorum*) is joined by a subspecies, the five-needled Arizona pine (*P. ponderosa* var. *arizonica*) (Kearney and Peebles 1951). Arizona pine normally dominates the lower elevational pine zones. It also can be found southward into the Sierra Madre where it is a major montane conifer and commercial tree (Pase and Brown 1982). Both Interior and Arizona ponderosa pines exhibit similar characteristics and growth behavior.

In the lower forest zone, as it grades into non-forest communities, ponderosa pine is typically a climax species. On higher, more moist sites ponderosa pine encounters more competition and becomes increasingly seral to Douglas fir and the true firs (Steele 1988). As a seral species ponderosa pine can dominate a site at some stage of succession. Historically, these stands developed as a result of low intensity surface fires that removed competing conifers and prepared a seedbed for pine (Cooper 1960, Steele 1988).

With its ability to put down a vigorous taproot, ponderosa pine has a high degree of drought resistance and is able to persist as a climax species on severe sites. Young seedlings are intolerant of shade but become moderately tolerant to competition as they move into the sapling and pole sizes. In Arizona and New Mexico, the climax ponderosa pine understory characteristically contains Arizona fescue (*Festuca arizonica*) and mountain muhly (*Muhlenbergia montana*) (Steele 1988).

Ponderosa pine reproduces well from seed with abundant seed crops every 2 to 5 years. Sexual maturity is reached in 15 to 20 years. Requirements for regeneration include an adequate seed source, exposed mineral soil seedbed, partial seedling shade, and adequate soil moisture (Conkle and Critchfield 1988). Fires aid regeneration by removing competing vegetation and preparing a suitable seedbed.

Other associated conifers commonly found in the Madrean ponderosa pine forest include Apache pine (*P. engelmannii*) and Chihuahua pine (*P. leiophylla* var. *chihuahuana*). Alligatorbark juniper (*Juniperus deppeana*) and evergreen oaks enter at the lower elevational boundaries (Pase and Brown 1982).

Apache pine has a limited distribution in southern Arizona and New Mexico but is common in the Madrean Region of Mexico. It is a codominant or dominant species, often occurring with Chihuahua pine, Mexican pinyon (*P. cembroides*) and alligatorbark juniper (Bowers and McLaughlin 1987). Also known as Arizona longleaf pine, Apache pine has unusually long needles that can range from 20 to 38 cm (8 to 15 in) in length. It can tolerate low light levels during establishment but becomes shade intolerant after about 6 years. Little (1950) reports that Apache pine seedlings pass through a grasslike stage, similar to the longleaf pine (*P. palustris*) of the Southeast. However, mature trees are more like the closely related ponderosa pines, differing mainly in the long needles, stouter twigs and slightly larger cones.

Chihuahua pine is commonly associated with Arizona and Apache pines at elevations ranging from 1,525 to 2,380 m (5,000 to 7,800 ft). This three-needled pine is a northern variety of a widely distributed Mexican species. It is quickly recognized by its many persistent, open, serotinous and semi-serotinous cones that remain attached to branches. Chihuahua pine is also one of the few pines that will sprout from cut stumps or from the root crowns (Little 1950). Seedlings are sensitive to strong light and heat and are more likely to establish beneath older trees, shrubs or near logs and boulders.

Mixed Conifer Forest

The mixed conifer forest is restricted to the highest mountain elevations where winters are cold and summers cool and moist. These forests are most extensive at the northern portion of the Madrean region (Felger and Johnson 1995). Most mature mixed conifer forests are dense, with high overstory canopies and heavy litter accumulations that restrict undergrowth (Pase and Brown 1982).

Niering and Lowe (1984) described the Madrean mixed conifer zone, found from 2,440 to 2,925 m

(8,000 to 9,600 ft) elevation, as a mix of species and habitats, including Rocky Mountain Douglas fir (*Pseudotsuga menziesii* var. *glauca*), white fir (*Abies concolor*), Engelmann spruce (*Picea engelmannii*), blue spruce (*Picea pungens*), southwestern white pine (*Pinus strobiformis*), ponderosa pine, and quaking aspen (*Populus tremuloides*).

Where Douglas and white fir occur together, white fir is considered to be climax. The two species join with spruce and southwestern white pine on cooler sites and with ponderosa pine on warm sites (Eyre 1980). At high elevations, especially on northern aspects, Douglas fir is replaced by Engelmann spruce and corkbark fir. Engelmann spruce often grows in dense stands, pure or mixed, with Douglas fir, white fir, and blue spruce (Little 1950).

Southwestern white pine, also known as Mexican white pine, occurs in low densities in the mixed conifer forest. Its best growth occurs on cool, moist sites with deep soils. Although southwestern white pine, a five-needled pine, is likely to be dominant at high elevations, it often persists as a long-lived seral species. When compared with associated conifers, southwestern white pine is relatively shade-intolerant, which may prevent its replacement as overstory canopies close.

Aspen, normally seral on moist sites, precedes the more tolerant spruce and fir species in the 2,125 to 3,050 m (7,000 to 10,000 ft) elevational zone. Virtually all aspen stands have developed by root suckering, often after a disturbance such as fire (Barnes 1975). Aspen has a lifespan of about 130 years and is shade intolerant. Without disturbance and with increasing age pure aspen stands change into an aspen-conifer mixture and ultimately to a pure conifer forest (Jones and Trujillo 1975). Aspen stands offer a rich wildlife habitat, providing abundant food and cover for a variety of birds and mammals (Patton and Jones 1977).

EFFECTS OF FIRE ON MADREAN MONTANE FORESTS

The effects of fire on Madrean montane forests are as diverse as the ecosystems themselves. Fires consume vegetative material, produce residual mineral products, and raise soil and air temperatures for short periods. In many fires, plant response depends more on the direct and indirect effects of higher temperatures than on available fuel or release of nutrients. Some heat effects may be immediate and easily observed, i.e., the killing of plant tissue; other effects may be delayed and more difficult to detect, such as increased insect and disease susceptibility or changes in species composition and distribution. This variability can cause problems for managers faced with the task of predicting changes in plant succession and ecosystem dynamics.

A plant's ability to withstand fire depends upon its heat tolerance and fire resistance. Heat tolerance is the ability of plant organs and tissues to endure elevated temperatures, whereas fire resistance is the ability of a plant to survive the passage of a fire (Zwolinski 1990). Both are related not only to fire intensity, duration, and weather conditions, but also to adaptations of individual plant species, i.e., bark thickness, sprouting, seed dispersal.

To obtain a better understanding of the effects of fire on the montane forests of the Southwest and subsequent management implications, a brief review of the impact of fire on the principal tree species of the ponderosa pine and mixed conifer forests is presented.

Ponderosa Pine Forest Species

Interior Ponderosa Pine

Historical evidence indicates that fires have always been an ecological force in ponderosa pine forests, regardless of whether they are climax or seral (Wright 1990). In Arizona and New Mexico, the natural fire frequency in climax and seral ponderosa pine stands varies between 4.8 and 11.9 years (Weaver 1951). Based on fire scars from a number of ponderosa pine sites in the Southwest, Swetnam (1990) reported a mean fire interval of 2 to 10 years between 1700 and 1900. After the turn of the century, fire intervals increased due to fire suppression efforts and removal of fire-spreading fine fuels by increased grazing. Subsequent large fires appeared to be linked to climatic fluctuations caused largely by El-Nino-Southern Oscillation events (Swetnam 1990). Ponderosa pine forests need frequent fires to maintain stand health and stability (Biswell et al. 1973).

Ponderosa pine has some species characteristics that minimize damage from fire. These include:

1. Thick, exfoliating bark,
2. Self-pruning lower branches,

3. Deep rooting habit,
4. Long needles loosely arranged in an open-crown structure,
5. High foliar moisture content, and
6. Ability to establish seedlings on fire-created seedbeds.

However, with an annual resinous needle production of over 3,000 lbs/Ac (3,360 kg/Ha), the species also can create flammable conditions by manipulating its own fuelbed (Biswell 1972).

Low intensity fires can readily kill seedlings less than 12 in (30 cm) high. Larger seedlings, saplings and poles can be damaged but, beyond pole stage, ponderosa pine is quite resistant to most fires.

Ponderosa pine must have an on-site seed source for regeneration, since seeds are too large to be distributed any distance. Seeds must also have a mineral seedbed so that the root radicle can be in contact with soil moisture. The big competitors to ponderosa pine seedlings are grasses for water and shrubs or small conifers for light. A fire in the first 3 to 5 years after seedlings are established can remove the pine (Wright and Bailey 1982).

The degree of crown scorch is considered a principal cause of pine mortality after fire (Dieterich 1979, Saveland and Bunting 1988). With its high foliar moisture content, ponderosa pine can withstand extensive scorching as long as buds and twigs, which can tolerate higher temperatures than needles, are not significantly damaged (Saveland and Bunting 1988). Some ponderosa pine trees can recover from scorch damage as high as 90 percent provided that 50 percent of the buds and twigs survive to maintain shoot growth on defoliated branches.

On moist sites, ponderosa pine often forms two-storied stands that may be quite susceptible to crown fire. The tendency of regeneration to form dense "dog-hair" understories on such sites creates fuel ladders that can carry surface fires to the overstory crowns (Biswell et al. 1973).

Arizona Pine

Since Arizona pine is closely related to interior ponderosa pine, adaptations and responses to fire are similar for both. The five needles per fascicle of Arizona pine provides an even more open crown arrangement and further discourages crown fires. Mature Arizona pines are also resistant to fire because of thick bark, deep rooting and length of branch-free trunk.

Plant communities historically associated with Arizona pine are fire dependent. Light, frequent surface fires in the past have suppressed shrubs, thinned Arizona pine stands, and created fertile seedbeds.

As with interior ponderosa pine, Arizona pine seedlings are very sensitive to fire, mortality of mature trees is related to crown scorch and bud/twig damage, and substantial amounts of needles are deposited each year.

Apache Pine

Responses to fire by Apache pine are not well documented in the literature. What information is available, principally from the Fire Effects Information System, a national computerized information storage and retrieval system (Fischer and McMurray 1990), indicates a close similarity of responses to fire as interior ponderosa and Arizona pines.

There is no clear evidence that the long needles and grasslike stage of Apache pine reported by Little (1950) has any close relationship to fire. However, it does appear that the reduced top growth and greater root development during this period would enhance survival of Apache pine on more xeric sites.

Chihuahua Pine

A major difference between Chihuahua pine and the other pine species of the ponderosa pine forest is the ability of Chihuahua pine to sprout from cut stumps and root crowns and to possess serotinous or semi-serotinous cones. This enhances the tree's ability to endure fire and regenerate on sites due to abundant seed production, delayed seed release, and sprouting potential.

Mixed Conifer Forest Species

Rocky Mountain Douglas Fir

Mature Douglas fir has a high resistance to fire damage. However, saplings and small poles are sen-

sitive to surface fires because of their thin bark, resin blisters, closely spaced needles, and thin twigs and bud scales. The low, dense branching habit of saplings and poles allows surface fires to carry into the crown. Older trees develop a thick, corky bark that protects the cambium against low to moderate intensity fires (Wright and Bailey 1982). Douglas fir foliage is considered to be highly flammable so even mature trees with branches extending the length of the bole are susceptible to "torching" into the crowns (Crane and Fischer 1986). As with ponderosa pine, heavy fuel accumulations at the base of the tree increase the opportunity for fire injury.

Douglas fir occurs in open stands, but it also grows in dense, continuous understories that will support a wind-driven fire. Even small thickets provide a path by which surface fires can enter the crowns of overstory trees. Crown fires are often assisted by the presence of lichens and witches' brooms caused by dwarf mistletoe (Alexander and Hawksworth 1975).

The shorter needles of many of the mixed conifer species, including Douglas fir, result in a compact, low porosity fuelbed. Slow moving surface fires with high residence times can remove the protective organic layer and damage shallow lateral roots.

Douglas fir regenerates on burned sites by wind-dispersed seeds. Fires will reduce fuel loadings and expose mineral soil allowing establishment of the shallow roots of seedlings. For best establishment Douglas fir needs minimum competition and some shade (Ryker 1975). Severely burned sites on south-facing slopes may be more favorable for ponderosa pine regeneration than Douglas fir because of the warmer, drier conditions.

Overall, Douglas fir is more fire resistant than spruces and true firs and equally or slightly less fire resistant than interior ponderosa pine. In the Southwest, frequent surface fires in dry Douglas fir habitats would maintain seral stands of ponderosa pine and/or southwestern white pine. Exclusion of fire will lead to the establishment of dense Douglas fir sapling thickets (Wright and Bailey 1982).

White Fir

Sapling and pole-sized white firs are fire sensitive. At this size trees have a smooth, thin, resinous bark and low-growing branches that can be easily ignited by surface fires. As trees mature the bark thickens and some self-pruning of lower branches occurs resulting in increased fire resistance. Shallow roots make white fir more susceptible to soil heating and root damage.

In the mixed conifer forest, a natural fire regime of frequent, low intensity surface fires prevents white fir from achieving climax dominance since it is less fire tolerant than associated species (Weaver 1951). This maintains the white fir habitat in a mid-successional stage where ponderosa pine or Douglas fir dominate the overstory and white fir exists in the understory. White fir will eventually become dominant on the site if the fire interval is long enough to allow trees to reach a fire-resistant size (Wright and Bailey 1982).

Engelmann and Blue Spruce

These spruces are very sensitive to fire and can generally be killed with low intensity burns. Postfire regeneration via wind-dispersed seeds readily takes place on fire-prepared seedbeds (Fischer and Bradley 1987).

Characteristics of the spruces that make them very susceptible to fire include:

1. Thin bark,
2. A moderate amount of easily ignited resin in the bark,
3. Shallow roots that can be damaged by soil heating,
4. Low hanging branches,
5. Moderately flammable foliage,
6. Tendency to grow in dense stands, and
7. Heavy lichen growth (Arno 1980, Crane and Fischer 1986, Fischer and Bradley 1987, Wright and Bailey 1982).

The high susceptibility of spruce to fire damage is mitigated somewhat by the moist and cool sites where it grows (Crane and Fischer 1986). Pockets of spruce can escape fire if they occur in wet locations where fire spread is hampered.

Southwestern White Pine

Not much fire effects information is available for southwestern white pine. It does not appear to be

well adapted to fire. Young trees with thin bark are sensitive to fire, while older trees with somewhat thicker bark become relatively more fire resistant. Horizontal or drooping branches increase its susceptibility to damage from fire.

Aspen

Aspen killed by fire will respond by vigorous root suckering and quickly form the dominant post-burn species on many sites. Fire removes apical dominance and, with a darker, warmer soil surface, stimulate roots to make stored food available for sucker generation. Moderate intensity fires appear most effective in promoting suckering (Bartos and Mueggler 1981). Aspen appears to be less susceptible to injury when burned or damaged during winter dormancy. Young sapling size or smaller aspen can also regenerate through root crown and stump sprouting.

Fire suppression and heavy grazing are contributing to a predominance of mature and decadent aspen stands in the Southwest. Most of these areas will require a major disturbance, such as fire, for their continuance (Barnes 1975).

FIRE MANAGEMENT IMPLICATIONS

Ponderosa Pine Forests

Many of the southwestern ponderosa pine forests are overstocked, stagnate, or maintain dense understories of young trees. These poor health conditions pose serious problems for managers. They were created because of lack of fire, however, recovery by restoring fire may also pose some difficulties (Wright 1990).

The management of pine forests by fire should recognize the vital attributes of the species present. As discussed earlier, these pines possess morphological and physiological characteristics that allow them to tolerate heat and survive the passage of all but the highest intensity fires. Managers who opt to bring fire back into these ecosystems need to do so incrementally to ensure that species tolerance for fire is not exceeded.

Prescribed burning can be used effectively in ponderosa pine forests, but it has to be site-specific (Harrington and Sackett 1990). The reduction of fuel accumulations is a major objective of fire use. A prediction equation for pine forest floor reduction based on humus layer moisture levels has been developed by Harrington (1987) from a series of summer burns in Arizona. Other elements of burning prescriptions for reducing fuels include site evaluations to determine fuel loadings, ladder fuels, potential fire intensity, energy release, and resistance to control. Fall versus spring or summer burning, fuel moisture, slope, and weather parameters also need to be considered (Biswell et al. 1973, Harrington and Sackett 1990).

Fire is also used in the ponderosa pine forest to reduce stand density and raise crown levels. Ignition techniques may involve lower intensity backing fires being met by more intense headfires that can create a vertical heat rise and energy concentration in the crowns (Harrington and Sackett 1990).

Ponderosa pine is considered a difficult species to regenerate in the Southwest due to moisture stress from droughts and competition from grasses (Pearson 1950). Prescribed fire is effective in removing forest floor material and grasses, thereby creating favorable seedbeds for regeneration.

Other beneficial effects of prescribed burning are increased production of herbaceous plants, improvement of habitats for wildlife species, increasing streamflow, and creating aesthetic environments (Ffolliott 1990).

Fire appears to be most beneficial for seral ponderosa pine stands since its role can be one of thinning, seedbed preparation, eliminating competition, and hazard reduction. These benefits aid in maintaining the pine in a seral stage. When fire is applied to climax ponderosa pine on its drier locations the smaller amounts of fuel/litter on the surface can be removed and bare soil exposed. Not only will protection from raindrops and soil movement be gone, but increased solar radiation also can create high soil temperatures that increase drought stress for new seedlings.

Harrington and Sackett (1990) indicate that prescribed fire in conjunction with different silvicultural treatments in the southwestern pine types has not received much attention. They point out dense slash following logging should be reduced to facilitate pine seedling establishment. However, it is known that large clearcuts followed by intense burning failed to promote regeneration of ponderosa pine seedlings (Schubert 1974). Harrington and Sackett

(1990) suggest that small group selection, shelterwood, or seed tree cuts followed by broadcast or pile burning will reduce fuel hazards and favor seedling establishment.

To keep the ponderosa pine ecosystem healthy and viable, stocking levels must be controlled. Prescribed fire, commercial and precommercial thinnings, and multi-product sales may be employed. In many areas fires can accomplish thinnings at a lower cost and be environmentally acceptable. A burning interval of 4 to 10 years would seem to be appropriate (USDA 1992).

Mixed Conifer Forests

Fires have played an important role in the composition and structure of the mixed conifer forests with ponderosa pine and Douglas fir the most resistant species. White fir, southwestern white pine, and aspen are intermediate in fire susceptibility. Least resistant and easily killed are Engelmann and blue spruces. Thus, Douglas fir and ponderosa pine usually are dominant while spruce is found in the understory. White fir and southwestern white pine are often fire-scarred, while patches of aspen are common, frequently marking the hot spots (Wright and Bailey 1982).

Mixed conifer forests lie at higher elevations where critical fire weather conditions are less frequent. Fuel loadings are higher in the mixed conifer type, averaging 98 metric tons/Ha (44 tons/Ac) versus 49 metric tons/Ha (22 tons/Ac) in ponderosa pine forests (Sackett 1979). Historically, wildfires were low to moderate intensity in mixed conifers, however, more stand-replacing crown fires can be expected in the future. Increased fuel loadings have resulted after decades of fire exclusion. Well-developed understories of shade tolerant conifers are common and will act as fuel ladders to crown fires. Fires used for fuel reduction will have to be handled carefully to avoid escape and stand damage (Jones 1974).

Jones (1974) presents the following as some possible uses of prescribed fire in mixed conifer stands: where an overstory of ponderosa pine or Douglas fir is to be maintained in lieu of more shade tolerant invading species, as a means of reducing fine fuels in fire-resistant stands using moderate burning conditions, for seedbed preparation after seed cutting in the shelterwood system, as a means of rejuvenating aspen stands for aesthetic and wildlife purposes where clearcutting is not feasible, as a sanitation measure against insects and diseases following partial stand cuts, and as a seminatural process in wilderness areas.

Periodic fire should be used in the lower zones of mixed conifer and intermingled ponderosa pine on a frequency of about 5 to 12 years, while within the higher pure mixed conifer type, fire at intervals of 20 to 25 years is recommended (USDA 1992). Pure mixed conifer stands contain closed canopies and are less tolerant of frequent fires. Fire use in the upper end of the mixed conifer zone containing scattered spruce should be infrequent. After fires, heavy fuel loads may remain on the site due to the high moisture levels in large fuels. In spruce dominant stands treatment of slash other than by prescribed fire may be needed to protect the residual stand (USDA 1992). The use of fire in this upper zone also can create openings for species diversity and wildlife habitat.

Fire appears to be the most economical management alternative for maintaining aspen stands. Fire suppression is considered to have greatly reduced aspen production (Bartos and Mueggler 1981, Jones 1974, USDA 1992). Although aspen may be difficult to burn at times, successful treatment can be achieved with adequate fine fuel loadings, continuous fine fuels, and cured herbaceous vegetation. Older aspen stands that are starting to break up also have increased flammability. Aerial ignition can enhance fire spread on marginal sites. Grazing by domestic animals and wildlife reduces fire potential, spread and intensity. Fencing a year prior to and after burning will help in preparing the site and protecting new aspen suckers (USDA 1992).

CONCLUSIONS

On a long term basis, the management of Madrean ponderosa pine and mixed conifer forests should emphasize the restoration of greater ecosystem stability and resiliency. Re-introduction of fire by carefully planned and executed prescribed burns is one option available for achieving this objective. However, the variability of fire and weather will always add an element of risk to this approach. The benefits of fire to these ecosystems, long adapted and dependent to some degree upon fire, must be weighed against the risks of prescribed burns escaping. Conversely, the same risks must be considered against

the increasing probability of sweeping catastrophic fires if no management options are exercised (Grissino-Mayer et al. 1995).

The past decade has provided a number of significant advances in our knowledge of and ability to predict fire effects and post-fire succession. A better understanding of adaptive characteristics of individual plant species and basic successional processes, coupled with the continued developments in fire behavior models and geographic information system data bases, is having an impact on fire management decisions (Zwolinski 1990).

But more information is needed. Managers and scientists must continue to prepare and analyze fire reports for prescribed burns and wildfires. Site variables, including fuels and weather conditions, at the time of burning need to be correlated with observed fire characteristics. Post-fire appraisal, both immediate and long term, must be made. As more information, under a variety of burning conditions, becomes available, managers will be able to open the window of burning opportunity wider and become more confident in restoring fire to the Madrean forest ecosystems.

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Fire Effects on Sonoran Grasslands

Dan Robinett[1] and Steve Barker[2]

Abstract.--Research on good-condition grasslands at Fort Huachuca, in Chochise County, Arizona, suggests that allowing fires to burn at "natural" intervals of 5 to 10 years does not diminish resource value or productivity. In fact, such fires may help keep ungrazed grasslands vigorous and healthy.

Wildfires played an important role in shaping grasslands in southeastern Arizona. (Pase 1977) Lightening in early summer storms or Indians using fire to hunt, clean up favored campsites or accidental starts from their campfires, were common prior to the turn of the century. (Bahre 1985 & Leopold 1924).

Natural fire frequencies for grasslands in southeastern Arizona have been estimated at between 10 and 20 years. (Wright 1980) A variety of cultural and environmental impacts since Anglo settlement have greatly reduced the frequency and spread of wildfires. (Bahre 1991) Over the last one hundred years very little of Arizona's grasslands have burned.

One of the few areas where fires have burned frequently is Fort Huachuca in Cochise County. Fort Huachuca is a US Army installation established in 1877 as an outpost in the southwest to quell Apache raiding. In 1916 it played an important part in General Pershing's punitive expedition into Mexico after Pancho Villa.(Wallmo 1951) Since then the 72,000 acre military reservation has grown to become the headquarters of the US Army Communications Command. It continues to accommodate troop training from regular and National Guard units in Arizona and from around the country. Several firing ranges and their associated impact areas occur on the 40,000 acre main post. These ranges are for live fire from troops and tanks. Wildfires caused by tracers are common each year.

This study started in 1992 to help determine how frequently fires can burn the major range sites without long term negative impacts to the soils or plant communities. This study gives some insight into the adaptation of the major species to fire (Robinett, 1994).

The grasslands around these training ranges are some of the finest in Arizona. Ecological condition is good to excellent in most areas. Although they have a long history of grazing in the past, most areas have not been grazed except by wildlife since 1948 when the last of the cattle were removed from the post. (Wallmo 1951) About 300 head of buffalo grazed on the main post until the early 1960s.

Elevations range from 4800 to 5400 feet in this area. Average annual precipitation is 16 inches. (NOAA 1992) Precipitation patterns produce two growing seasons. Cool season moisture tends to be frontal storms with moisture supplies from the Pacific and summer rainfall comes as convective storms of high intensity and short duration from moisture supplies originating in the Gulf of Mexico.

Fire history data has been kept on Post since 1977. The extent at each burn was delineated on a training range map along with the time, dates and a brief explanation of how it started and how it burned. Using this data and a recent soil and range survey of the Post, sampling areas were selected. (USDA 1992)

A combination of three different fire frequencies on the four major range sites in the area were evaluated. The three burn frequencies are; one burn since 1977, two or three burns since 1977, and five or six burns since 1977. All fires were in the hot season of May through July. The one burn areas had from 6 to 8 years since the burn; the two and three burn areas had from 4 to 6 years since their last fire; and the last burn on the five and six fire areas was in 1990 or 1991.

The four range sites sampled included ; Loamy upland, Sandyloam upland, Loamy hills, and Granitic hills. (USDA 1988)

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The loamy upland range site is characterized by deep soils classified as ustollic haplargids and haplustalfs. (USDA 1992) They have thin (1 to 3 inches), coarse textured surfaces over clayey subsoils. Slopes are 1 to 5%. The native potential plant community is an open grassland dominated by warm season mid-grasses.

The sandyloam upland range site is characterized by similar soils but with a thicker (4 to 10 inches), coarse textured surface. Slopes are 1 to 3% and the potential native plant community is similar to Loamy upland except production is higher.

Both of these sites were open grassland in the early day photos of Fort Huachuca but now have a light to moderate cover of velvet mesquite.

The loamy hills range site has deep soils classified as ustollic paleargids or haplargids and argiustolls and paleustalfs. (USDA 1992) They have thick (8 to 16 inches), very cobbly and gravelly, dark colored, sandyloam surfaces over dense clay subsoils. Slopes are from 10 to 35% . The potential native plant community is a grassland with a moderate percentage of low shrubs and succulents.

The granitic hills range site has shallow soils classified as lithic haplustolls and lithic argiustolls. (USDA 1992) They have very cobbly and gravelly surfaces, are dark colored loams and sandyloams over slightly weathered granite bedrock. Slopes are from 20 to 50%. The native potential plant community is savannah with a 10 to 25% canopy of Mexican live oaks and an understory of warm season mid-grasses, perennial forbs and low shrubs.

One sampling area was selected for each of the range site-burn frequency combinations. Site selection was heavily biased to represent what appeared to be average conditions for the area being studied. This was not a research effort. It was an investigation designed to produce some information about fire effects in a short period of time with a reasonable amount of effort.

Transects used different techniques to measure different attributes of the plant community. Basal, rock and gravel cover were measured as line intercept along three 100 foot steel tapes. Canopy cover was measured as shaded line intercept along the steel tapes at mid day. Frequency data was collected using a 40 square centimeter quadrat in a 100 plot transect. Plant composition data was derived using the same quadrat size and transect and the Dry Weight Rank method. (Ruyle 1988) Plant production data also used the same transect and quadrat size and the Comparative Yield method. (Ruyle 1988)

LOAMY UPLAND

This site appears to be the most affected by repeated fires. This site naturally produces a lot of runoff in the summer rainy season. The thin , coarse textured surface cannot capture all of the larger rainfall events. If the surface is not protected by grass and / or gravel cover, accelerated erosion can begin. Basal cover, annual herbage production and number of plant species all declined as fire frequency increased (fig. 1).

Basal cover on the one burn and three burn sites was about 15% and no erosion was evident. The site that had burned five times in the last 15 years had only 6.5% basal cover and visible signs of accelerated erosion. As this site looses its surface horizon to water erosion it becomes less effective in capturing and storing intense summer rainfall. With enough soil loss , the potential productivity declines and the site can no longer support its natural plant community.

Another observation on this site was that even with 5 fires in the last 15 years there was little or no mortality of mature velvet mesquite trees. It appears that, at these elevations (4800 ft.) and with 16 inches

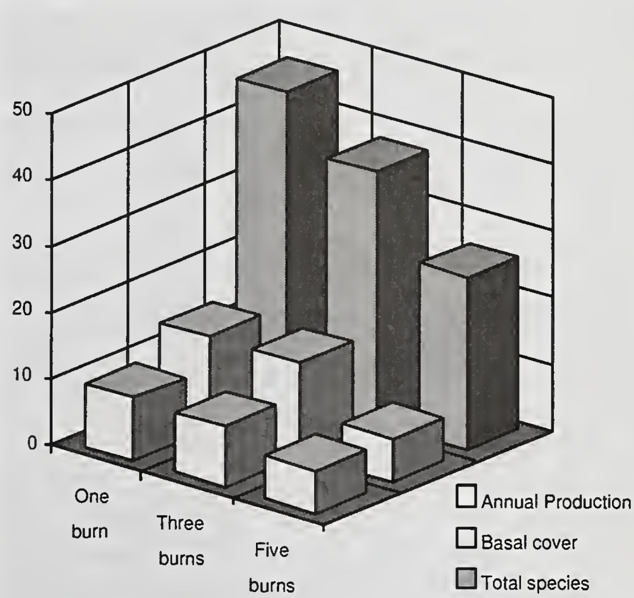

Figure 1. Cover, production and total species on Loamy Upland range site

of annual precipitation, established mesquite trees can survive a very frequent summer fire regime.

SANDYLOAM UPLAND

This site , with thick coarse textured surfaces, produces very little runoff in the summer rainy season. Even with repeated fires this site showed no signs of accelerated erosion. Basal cover was nearly the same for all three burn frequencies. This site is the one most favored by Lehmann lovegrass in southeastern Arizona. Lehmann lovegrass is a warm season , perennial bunchgrass , introduced into this area from southern Africa in the 1930s. Since then it has steadily spread across southeastern Arizona developing into nearly monotypic stands on this range site.

Grazing, fire and drought have been implicated in the invasion of native grasslands by this species. (Ogden 1988) The opportunistic nature of this species to respond to openings in native plant communities caused by fire (Cable 1965, 1971) and drought (Robinett 1992) has been documented in this region. Frequency of Lehmann lovegrass went from 9% on the one burn site to 96% on the five burn site. This was at the expense of sideoats and black gramas and plains lovegrass (fig. 2). Although annual production and cover remained nearly the same among the three burn frequencies , total number of plant species declined from 47 on the one burn area to 29 on the five burn area dominated by Lehmann lovegrass.

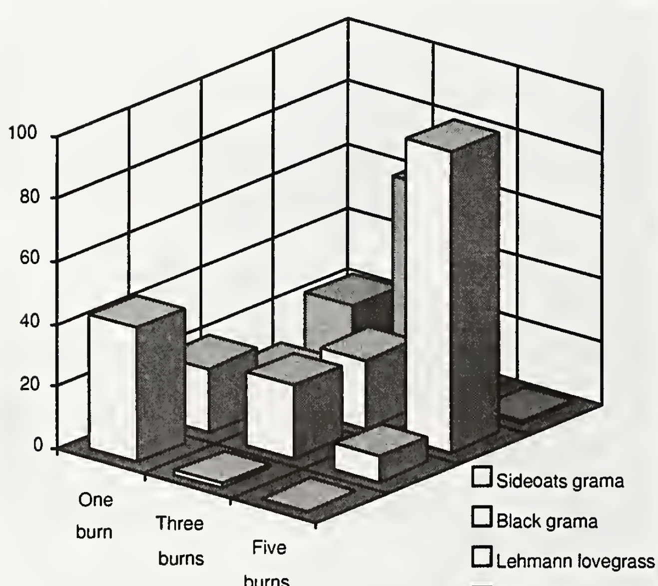

Figure 2-Frequency of grass species on Sandyloam Upland range site

The ability of mature velvet mesquite trees to withstand frequent fires was noted again on this site. Another observation was that there was considerable decadence of native grass species ; sideoats grama, cane bluestem and plains lovegrass in the one burn area (fire in 1984). Correspondingly there was a much higher percentage of annual forbs like goldeneye and aster in the plant community on the one burn area.

LOAMY HILLS

This site has thick coarse textured surfaces, well protected by covers of stones , cobbles and gravels. Even on steep slopes, sites with frequent fires showed no sign of erosion. Basal cover, total number of species and production were the same for all three burn frequencies. Differences in species composition are within the range of variability for major grass groups in the potential plant community on this range site (table 1). Midgrasses like sideoats grama, Arizona cottontop, tanglehead, cane bluestem and green sprangletop are included in a group at 20 to 30 percent on the published range site description. (USDA 1988) Range condition scores on the three areas ranged from 70 to 72.

Surface rock fragments on this site not only protect the soil from accelerated erosion but also appear to

Table 1. Species Composition on Loamy Hills range site

	One burn	Three burns	Five burns
% Plain lovegrass	22	24	40
% Lehmann lovegrass	0	1	2
% Tanglehead	32	6	3
% Sideoats grama	6	13	8
% Slender grama	1	3	10
% Cane beardgrass	6	8	2
% Green sprangletop	8	5	4
% Fall witchgrass	1	4	10
% Threeawns	4	10	3
% Arizona cottontop	0	13	1
% False mesquite	11	11	16
% Palmer agave	4	2	4
Ann. production lbs/ac	2375	2151	2360
Basal cover%	15	15.5	16
Cobble cover%	14	16	18
Total plant species	34	37	32

protect the bases of perennial mid grasses from damage by hot season fires.

This site is a primary habitat for Palmer agave in southeastern Arizona. The blossoms of this agave are a major food source for a nectar-feeding bat which is listed as an endangered species. The lesser long-nosed bat uses saguaro and organpipe flowers in May and June and agave flowers in July and August during its migration from tropical Mexico to Arizona each year. Seedling agave plants are easily killed by hot season fires while older plants appear fire tolerant. One visible difference in the five burn plot on this site is that there are no carcasses of dead agave plants left on the area. Palmer agave lives 15 to 25 years, flowers and dies. The large heavy seeds fall from the panicle straight down and a high proportion of seedling establishment occurs around the base of the dead adult carcass. Frequent fires consume the dry dead plants and intense heating kills any seedlings growing nearby.

Again it was noted there was considerable decadence among mid grasses like tanglehead, plains lovegrass and sideoats grama on one burn area (burn in 1984). This was not noted in the other burn areas.

This phenomenon has been noted by others working in similar grasslands. These midgrass communities become decadent in the abscence of fires especially during short regional droughts like those in 1988-89 and again in 1993-94. Openings in the plant community are quickly occupied by annual forbs like goldeneye and camphorweed which can be very competitive to grasses in mild, wet winters. Spring or summer fires kill the forb seedlings, remove litter and rejuvinate the perennial grass plants. On the Research Ranch, plains lovegrass declined significantly in abundance in unburned areas due to mortality in the 1988-89 drought. It did not suffer similar mortality on adjacent areas that had burned in 1984 or in 1987. (Bock and Bock, et.al. 1995) In another study on the Gray Ranch in southwestern New Mexico, it was shown that a summer fire killed annual goldeneye and greatly reduced its seedlings the next year. This would reduce competition to perennial grass plants recovering from drought. (Sundt and Turner, 1994)

GRANITIC HILLS

This site has shallow , coarse textured soils well protected by covers of stones, cobbles and gravels. Even on very steep slopes the only area showing signs of accelerated erosion was the 6 burn hillside. The area with three burns in the last 15 years showed no signs of erosion and illustrates the effectiveness of rock fragments and grass cover in protecting the soil and the remarkable adaptations of dominant grasses like Texas bluestem, plains lovegrass, bullgrass and sideoats grama to frequent fires.

Basal cover, annual production and total number of species were not different between the three burn frequency areas of this site. (Table 2) One visible difference was in the crown canopy of oak species found on the site. The 6 burn area had about half the tree canopy of the 1 burn area. These species of evergreen oak are very fire tolerant and vigorous sprouters but a few dead individuals were present on the 6 burn area and the repeated burning appears to prune the tree canopy and reduce it's lateral extent. As expected shade tolerant understory species like sedges and stolon daisy were much more common in the 1 burn area with double the canopy of the 6 burn area.

Protecting areas of this site from fire for very long periods of time can lead to thickening of the tree cover to the point where herbaceous understories are greatly reduced. This appears to have happened in the last hundred years in the mountain ranges nearby where grazing reduces fine fuels and protection from fire occurrs. (Humphrey 1987) A fire history study done nearby in the ponderosa pine forest atop Ramsey peak showed a mean fire return interval of 8.5 years

Table 2. Species Composition on Granitic Hills range site

	One burn	Three burns	Six burns
% Texas bluestem	22	21	19
% Plains lovegrass	5	11	16
% Beggartick threeawn	24	13	6
% Sideoats grama	3	25	4
% Bullgrass	0	3	15
% Squirreltail	5	0	2
% Sedge	16	4	8
% Oak species	4	3	3
% Wedgeleaf haplopappus	3	0	0
% Wild bean	4	1	2
% Herbaceous sage	1	8	8
% Stoloniferous daisy	6	2	0
Ann. production lbs/ac	938	1214	1098
Tree canopy cover%	23	13	10
Basal cover%	8	12.5	9.5
Rock / cobble cover%	16	15	16
Total plant species	57	51	47

for the period 1700 through 1899. (Danzer, 1995) A comparison of photographs from the 1880's taken on Fort Huachuca in areas which have not burned in the last 30 or 40 years, show a thickening of the tree cover in present day scenes. When areas like this do eventually burn, erosion can be serious because there is insufficient grass cover to hold soils in place.

A study done nearby on similar soils on the Empire Ranch used rainfall simulation on burned and control plots to detect differences in runoff and sediment production. Immediately after the first fire there was no differences in either runoff or erosion between the burned and the unburned plots. (Emmerich and Cox, 1992) There was significantly greater runoff and sediment production from the burned areas one year later after a second fire. (Emmerich and Cox, 1994) These studies indicates the negative effects of fire regime in that these grassland soils retained their structure for some time after one fire, minimizing runoff and erosion. But after one year, soil structure was lost producing increased runoff and erosion. Three years after the burns, runoff was still elevated, while erosion had returned to pre-burn levels.(Emmerich, 1996)

The general information resulting from the study should be of interest to land users and managers in nearby areas. Allowing fires to burn these plant communities at what is thought to be natural intervals of 5 to 10 years does not appear to diminish resource values or productivity. It may actually be beneficial on many sites to keep ungrazed grasslands vigorous and healthy. A followup to this study is presently being done by the US Geological Survey. This research assesses the nutrient status of the soils in this area under the different fire regimes on Fort Huachuca. Preliminary results indicate that a moderate fire regime (3 fires since 1977) results in higher nutrient status in these soils (C, N, P) than either no fires or a frequent fire regime (6 fires since 1977). (Biggs et. al., 1966)

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Effects of Fire on Riparian Systems

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Abstract.—Riparian systems are a small but important resource in the southwestern USA and northern Mexico because of the diverse, dynamic, and complex biophysical habitats they provide. Wildfires have always produced the most significant impacts on riparian hydrology, geomorphology, and biology. Prescribed fires have not been used to any great extent in the Madrean Province for vegetation management in riparian systems. The unique geophysical characteristics of the Madrean Archipelago make it especially responsive to climatic events. Water yield, peakflow, and sediment yields after wildfires can be among the largest in North America, and can have a substantial impact on the watershed and associated riparian systems.

INTRODUCTION

Fire has occurred regularly in riparian and wetland environments for thousands of years. During the Holocene period, native Americans used fire in their hunting efforts to drive large mammoths into streams where they slew them for food (Dobyns, 1989). Later, hunter-gathers set fires to remove dead biomass from pond-marsh wetlands to improve game animal habitat and game bird productivity.

Today, both wildfires and prescribed fires occur in many riparian and wetlands systems. In riparian systems, fire is usually in the form of an unintentional invasion during a wildfire. Little prescribed fire in being used in the riparian systems for their management. However, some prescribed responses of fire in riparian systems can be gleaned from the experience with wildfires.

This paper reviews the limited studies reported on the effect of fire on riparian systems in the southwestern United States. This information is then used along with information from studies reported in other parts of the United States to assess the role of prescribed fire in riparian systems in the Madrean Province.

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RIPARIAN SYSTEMS IN THE SOUTHWEST

Riparian systems are found in a wide range of elevations throughout the southwestern United States and Mexico. Their classification is complicated by variety of physical and environmental factors that have contributed to high plant species diversity. One classification system places riparian ecosystems into two general cover types - forest and scrub (Szaro, 1989). The riparian forests are further classified by different elevations, namely above 2000 m, below 2000 m, below 1500 m, and below 1000 m. Distinct scrub riparian areas have been broken into two categories, those above 1500 m and those below 1500 m. This classification scheme does not include marshlands and cienegas which can occur at both lower and higher elevations.

IMPORTANCE OF RIPARIAN SYSTEMS

Natural riparian systems are considered the most diverse, dynamic, and complex biophysical habitats in the Southwest. Because riparian systems act as interfaces between terrestrial and aquatic systems, they possess sharp environmental gradients, ecological processes, and biotic communities. Therefore, riparian ecosystems represent an unusually diverse mosaic of landforms, communities, and environments within a larger landscape setting. As such, they serve as a framework for understanding

the organization, diversity, and dynamics of communities associated with fluvial systems (Naiman et al 1993). Under most conditions, riparian plant communities are best left alone or even protected from logging, grazing, and other types of exploitation. Unfortunately, the response and role of fire in riparian system dynamics is not well understood (Baker 1990).

Healthy riparian systems provide values and benefits far in excess of the land area they occupy. Healthy riparian systems provide water and soil to encourage vegetation growth for a more productive biotic community which has high biodiversity. They support a large diversity of insect, mollusk, and crustacean species that are key resources in the food chain. Healthy riparian systems also provide amenities for people by creating cool shade and general aesthetic beauty with their serene waters; and they provide prime areas for fishing, hiking, rafting, bird watching, water sports, picnicking, and camping. The lush vegetation improves water quality and removes sediment; rebuild flood plains and reduces erosion of stream banks; holds water in the streambanks which improves local ground water reserves; maintains instream biota; and stores flood waters, lessening the risk of flash floods (DeBano and Schmidt 1989).

ROLE OF FIRE IN RIPARIAN SYSTEMS

Riparian ecosystems are corridors streambank vegetation which interact continuously with the surrounding watershed, and represent a reservoir of disturbance-oriented species within the confines of a less frequently disturbed landscape. The edges of riparian ecosystems are often buffered from upland disturbances such as fire and to some extent, fluvial disturbances. Typical mid-sized riparian ecosystems, therefore, exhibit decreasing disturbance toward their core by processes common to uplands (e.g. overland flow,surface erosion, etc.) and increasing disturbance by fluvial processes (e.g. channel aggradation and degradation). Fire can affect both the channel processes in the riparian ecosystem and hillside processes on the surrounding watershed.

Both wildfires and prescribed burns can affect riparian systems directly and indirectly. If fires burn in the riparian area itself, direct effects such as the consumption of part or all of the vegetation is immediately obvious. However, subtle indirect effects such as stream temperature increases, alterations in the quantity and quality of organic matter inputs to streams, aquatic macroinvertebrate population changes, and fish migration can occur long after the fire. When fire burns the surrounding watershed, direct effects such as excessive surface runoff, increased streamflow, higher peakflows, and sediment movement into and through downstream riparian ecosystems are readily apparent. Nutrient cycling changes, plant and animal community shifts, population declines, and other indirect effects can occur but are not easily visible.

Fire decreases basin stability on steep topography, debris flows along with dry ravel and small landslides are a common occurrence after fire. The recovery of vegetation following fire reflects the combined disturbance of both fire and flooding. The direct effects of fire on riparian ecosystems consists mainly of damage to the vegetation (trees, shrubs, and grasses) and partial consumption of the underlying litter layer. The severity of the damage depends largely upon the intensity of the fire. Intense wildfires can cause severe damage to plant cover as contrasted to low intensity cool- burning prescribed fires which have less severe consequences.

Both wildfires and prescribed fires on the watershed affect the downslope riparian ecosystems indirectly by changing the fluvial processes on a watershed. The most obvious change involves the removal of protective vegetative and litter cover which intercepts precipitation. When the protective soil cover is removed, the soil surface becomes subjected to raindrop impact and as a result both surface and rill erosion can increase. The increased erosion is related to the amount of protective cover removed. The effects of fire are variable and depend partly on the severity of the fire. Cool burning prescribed fires have little impact whereas intense wildfires may have substantial impact on the stormflow, erosion, sedimentation, and quality of the streamflow. The duration of these effects is affected by the rate of post-fire revegetation.

RESPONSE OF RIPARIAN SYSTEMS TO FIRE

The biotic communities that make up the Madrean Province floristic province consists of montane coniferous forests, oak-pine woodlands, tropical de-

ciduous forest, savanna, short-grass prairie, subtropical thornscrub, and subtropical desert (Brown 1982; Warshall 1995). The specific responses of these Madrean ecosystems to fire have been reasonably well-studied in northern end of the Province (central and southern Arizona), but are limited in the southern end (Sierra Madre Occidental). Information from other better studied but comparable floristic zones (Californian oak-woodlands and chaparral) are used in this discussion.

Vegetation

Impacts on riparian vegetation can range from small effects with prescribed fires to severe effects associated with wildfires. The recovery of vegetation following fire reflects the combined disturbance of both the fire and any subsequent flooding. In southern California, post-fire recovery processes in the herbaceous layer were closely linked to geomorphic location in the riparian zone, and to the density of seeded, non-native ryegrass (Davis et al. 1989). Annuals became well-established on higher geomorphic locations less prone to flooding, but often in loose soil subject to dry ravel. Perennials, on the other hand, grew better on lower, more disturbed geomorphic locations near the stream. The overall species richness of annuals decreased in the second year after fire due to the redominance of ryegrass, although perennials took over the riparian area to a large extent. Sprouting is the dominant means of recovery of the tree species due to the lack of viable seeds following fire. Species such as sycamore (*Platanus racemosa*) showed rapid recovery, while others such as alder (*Alnus rhombifolia*) were slow to reestablish in the absence of a viable seed source. Full recovery of the alder canopy after unusually hot fires can take many years or decades.

Some riparian areas recover rapidly following wildfires, whereas others undergo slow recovery. The rate of recovery depends largely on the environment. A study on Marble Cone fire near Carmel, California after a intense wildfire showed that large amounts of sediments were deposited in riffle areas immediately following the fire (Hecht 1984). However, repeated measurements following the fire showed that habitat values of riffles were largely restored by vegetative growth by the end of the first year, and had undergone virtually complete recovery after three years. In contrast, fire in desert riparian environments might show prolonged effects. Although fire is not a common occurrence in desert riparian ecosystems, the riparian communities can be ignited by rolling firebrands from the surrounding desert grasslands (Miller et al. 1995). Although many riparian plant species can re-establish after fire, recovery of the vegetative structure may take a long time.

In southeastern Arizona, the response of big sacaton (*Sporobolus wrightii*) grassland riparian sites to burning and mowing were studied Arizona (Cox and Morton 1985). Both burning and mowing reduced green biomass production, and stocking rates on the burned and mowed pastures were only one-third as high as on untreated areas. Cattle gains were also less on the treated sites.

Water Yield

Water yield increases from prescribed fires and wildfires in Madrean-type ecosystems are shown in Table 1. Where soil wettability becomes a problem, water yield increases can be high due to a greater contribution of surface runoff to stormflows. Overland flow that is usually low in most undisturbed forests can increase to 15-40% of total water yield after fires. If more precipitation leaves a watershed as surface runoff, baseflows will eventually decrease. Perennial streams become ephemeral in extreme conditions. The effect on biota in riparian and aquatic ecosystems then becomes devastating. Also, water supply for human use is impaired.

Peakflows and Flooding

The effects of fire on storm peakflows are variable and complex (Table 2). Storm peakflows can produce some of the most profound impacts that forest managers have to consider. The high velocities and flow volumes of peakflows are responsible for most sediment transport, and alteration of channel geomorphic characteristics. These flows also have profound influences on riparian biota. Thus, there has always been considerable concern that increases in annual flood peaks of 20+% could lead to channel instability and degradation, aquatic and riparian habitat deterioration, and increased property damage in flood-prone urban areas.

A wildfire in Arizona ponderosa pine produced a peakflow 58 times greater than an unburned watershed during record autumn rainfalls where the burn

Table 1. Increased Water Yield From Burned Watersheds In Madrean-type Ecosystems.

Watershed/ condition	Rain (mm)	Flow (mm\yr)	Increase (%)	Recovery (years)	Reference
AZ Chaparral	740				Davis (1984)
Control		64	—		
Burned		156	144	>11	
AZ Chaparral	655				Hibbert (1971)
Control		0	—		
Wildfire		124	9999+	9+	
Control		19	—		
Wildfire		289	1421	9+	
AZ Pinyon-Juniper	480				Hibbert et al (1982)
Control		34	—		
Burned		39	12	5	
TX Juniper-Grass	660				Wright et al. (1982)
Control		2	—		
Burned		25	1150	5	
Burned, Seeded	10	400	2		

was severe (Table 2). Watersheds in the Madrean Province are much more prone to these enormous peakflow responses due to geomorphology (high elevation ranges), climatic (monsoon weather conditions and a close source of moist, tropical air), and soil (shallow, clay-rich, and potentially hydrophobic) conditions (DeBano 1981; Swanson 1981).

Another concern is the timing of stormflows or response time. Burned watersheds respond to rainfall faster, producing more "flash floods". Hydrophobic and bare soils, and cover loss will cause flood peaks to arrive faster and at higher levels. Flood warning times are reduced by "flashy" flow and higher flood levels can be devastating to property and human life. As indicated in Table 2, the Southwest and the Madrean Archipelago is particularly vulnerable to changes in peakflow response time and volume. The responses are an interaction between intense rainfalls common in the region, steep terrain, and water repellency which often develops in shrubland vegetation typically found in the Madrean Province. Another aspect of this is the fact that recovery times can range from years to many decades.

So, the net effect on watershed systems and aquatic habitat of increased peakflows is very much a function of the type of fire, area burned, climate, watershed and soil characteristics, and the severity of the fire. Small areas in flat terrain subjected to prescribed fires will have little, if any, effect on water resources, especially if Best Management Practices are utilized. Peakflows after wildfires that burn large areas in steep terrain can produce significant impacts.

Table 2. Effects of Fire on Peakflows in Madrean-type Ecosystems.

Location	Treatment	Peakflow Control $m^3/s/km^2$	Peakflow Burn $m^3/s/km^2$	Reference
Chaparral, CA	Wildfire	0.051	1.169	Nasseri (1989)
Chaparral, AZ	Wildfire	0.008	7.800 a	Hibbert (1985)
		0.035	1.780 b	
Ponderosa Pine, AZ	Wildfire-Mod.	0.070	0.240	Campbell et al. (1977)
	Wildfire-Sev.	0.070	4.067	
Ponderosa Pine, AZ	Wildfire	0.060	1.400	Rich (1962)

a Summer storms; b Winter storms

Sediment

Total Yield

Some sediment yield baseline data are discussed by Neary and Hornbeck (1994). Natural erosion rates for forests in the western USA (0.001 - 5.530 Mg/ha) are higher than the East (0.110 - 0.220 Mg/ha) but do not approach the upper limit of geologic erosion (15.000 Mg/ha). Landscape disturbing activities like mechanical site preparation, agriculture, and road construction produce the most sediment loss and can match or exceed the upper limit of natural geologic erosion.

Sediment Yields From Fires

The regions where fire accounts for the highest portion of total sediment yield in North America, include the Madrean Archipelago and the chaparral steeplands of Southern California (Swanson 1981). Sediment yields are usually higher immediately following a fire and then decline rapidly in subsequent years as vegetation is reestablished. All fires increase sediment yield, but it is the wildfires that produce the biggest amounts (28 - 369 Mg/ha). Slope plays a major role in determining the amount of sediment yield, but *Best Management Practices* are useful in reducing sediment loss in steeplands (Heede 1988).

Debris Flows

Observations in southern California show that wildfires (e.g. Wheeler Fire) decrease basin stability. In steep erodible topography, debris flows along with dry ravel and small landslides off hillslopes are a common occurrence (Davis et al 1989). Debris flows are the largest, most dramatic, and main form of mass wasting which delivers sediment to streams. They can range from slow moving earth flows to rapid avalanches of soil, rock, and woody debris. Debris avalanches occur when the mass of soil material and soil water exceed the sheer strength needed to maintain the mass in place. Steep slopes, logging, road construction, heavy rainfall, and fires all aggravate debris avalanching potential.

Most mass failures after fire are associated with development of water repellency in soils. Chaparral and other sclerophyllous vegetation in the southwestern USA and the Madrean Archipelago are high debris flow hazard areas because of the tendency for water repellency to develop in soils of these ecosystems after fires (DeBano 1981). Debris flows can be 50% of the total post-fire sediment yield in some ecosystems. In Southern California chaparral, sediment yields from debris flows increased from 7 to 1907 m3/km2/yr the first year after a wildfire (Wells 1981).

The impacts of debris flows on riparian ecosystems are multiple. Large, sudden inputs of sediments into streams cause rapid aggradation of channels. A stable stream channel reflects a dynamic equilibrium between incoming and outgoing sediment and streamflow. Increased erosion after fires can alter this equilibrium by transporting additional sediment into channels (aggradation). Although increased peakflows that result from fires can also produce channel downcutting (degradation), the process of aggradation from debris flows dominates in the short-term. Narrow, distinct channels turn into broad, braided systems. The consequences are destroyed aquatic habitat, damaged riparian vegetation, devastation to aquatic biota, deteriorated water supply systems, and spreading and elevating of flood flows.

Buffer Strips

Riparian systems along rivers in the Madrean Province serve an important function as buffer strips which capture sediment and nutrients, thereby preventing them from entering the stream. Buffer strips are protective areas adjacent to an area requiring special attention. Filter strips are buffer strips specifically designed to trap sediment.

An important part of developing any prescribed burning program in riparian areas is recognizing their importance as buffer areas for adjacent streams. Guidelines on buffer strip widths for prescribed fires have been established for other parts of the United States. For low intensity fires, less than 0.6 m high that does not kill stream-shading shrubs and trees, fire can be used throughout the riparian area without creating substantial damage (Neary et al 1993). Where fire damages woody vegetation, the width should be proportional to the size of the contributing area, slope, cultural practices in the upslope area and the nature of the drainage below (Cooper et al 1987). Buffer widths should increase as higher order streams

are encountered and deposition opportunities within the zone decrease. A general rule of thumb for the width of the buffer strip which should be planned for is 9 meters plus (0.46 X % slope) (USDA Forest Service 1989).

Large Woody Debris

Large organic debris is recognized as an increasingly important component of watersheds and river systems in the western United States. Large woody debris plays an important role in hydraulics, sediment routing, and channel morphology of streams flowing through riparian systems (Smith et al. 1993). Woody debris increases the complexity of stream habitats by physically obstructing water flow. Trees extending partially across the channel deflect the current laterally, causing it to widen the streambed. Sediment stored by debris also adds to hydraulic complexity, especially in organically rich channels that are often wide and shallow and posses a high diversity of riffles and pools in low gradient streams of alluvial valley floors. Stream stabilization after major floods, debris torrents, or massive landslides is accelerated by large, woody debris along and within the channel. After wildfire, while the post-fire forest is developing, the aquatic habitat may be maintained by large, woody debris supplied to the stream by the prefire forests.

Managers often debate the post-fire management practice of leaving burned and dead debris in channels following wildfires. The USDA Forest Service suggests that the *Best Management Practice* balances downstream value protection with the environmental implications of the treatment (Barro et al, 1988). To protect life and property, treatments generally involve channel clearing, necessarily at the expense of the riparian environment. Because living and dead trees in the riparian zone act as stabilizing elements in streambed configuration, their removal can provoke adjustments in channel morphology. In addition, removal of obstructions increases flow velocity which can scour the channel bed, increase the sediment load, degrade the water quality, export nutrients out of the system, and cause deterioration of the biotic habitat. Therefore, in the final analsysis it appears beneficial to maintain as much woody material as is possible in the ecosystem following fire.

Water Chemistry

Undisturbed forest, shrub, and range ecosystems usually have tight cycles for major cations and anions, resulting in low concentrations in streams. Fire interrupts or terminates uptake by vegetation and speeds up mineral weathering, element mineralization, microbial activity, nitrification, and decomposition. These processes result in the increased concentration of inorganic ions in soil solution and leaching to streams via subsurface flow.

Nutrients carried to streams can increase the growth of aquatic plants, reduce the potability of water supplies, and produce toxic effects. Anions like phosphate and cations such as calcium and potassium can be exported from watersheds at 10 times their normal rate immediately after severe disturbances, but do not significantly alter water quality.

Most of the attention relative to water quality after fires focuses on nitrate nitrogen (NO3- N), since it is highly mobile. High NO3-N levels in conjunction with phosphorus can also cause eutrophication of lakes and streams. Most studies of forest fires show increases in NO3-N concentrations with maximums in Madrean-type ranging from 0.6 to 12.0 mg/L. Losses of NO3- N would be higher if volatilization into the atmosphere was not a major pathway of nitrogen loss. Except for the municipal water quality implications of elevated NO3-N concentrations, water chemistry changes in Madrean Province ecosystems have not been a problem.

Temperature

Large fires can function like clearcuts in raising the temperature of streams due to direct heating of the water surface by solar radiation and increases in excess of 150 C have been measured .

The main concern relative to riparian ecosystems is the reduction in the concentrations of dissolved oxygen (O_2) available for aquatic biota that occurs with rising temperatures. Dissolved O_2 contents are affected by temperature, altitude, water turbulence, aquatic organism respiration, aquatic plant photosynthesis, inorganic reactions, and tributary inflow. Generally, O_2, concentrations <10 mg/L create problems for fish. Increases of 1-5 C that are not a problem at sea level become problematic at high altitudes of the Madrean Province .

Aquatic Macroinvertebrates and Fish

The most sensitive species in riparian ecosystems to fire effects are aquatic macroinvertebrates and fish. Rinne and Neary (This Volume) cover this topic in some detail so no further discussion is warranted here.

CONCLUSIONS

Fire has not been used extensively in management of riparian and wetland systems in the Madrean Archipelago region. Most fire in riparian areas occurs in the form of an unintentional invasion during a wildfire. Under these conditions, fires in the riparian area can be intense and cause extensive damage to the vegetation. However, even after severe fires, recovery to prefire conditions can be rapid—within a couple years in some environments. The recovery of vegetation following fire reflects the combined disturbance of both the fire and flooding.

Because of the high responsiveness of Madrean Province watersheds to climatic events, water yields, peakflows, and sediment yields measured after wildfires have historically been some of the largest in North America. Thus, the potential is always there for significant adverse impacts to riparian ecosystems and the biota that inhabit them.

Riparian areas provide buffer strips which trap sediment and nutrients that are released when surrounding watersheds are burned. The width of this buffer strip is critical for minimizing sediment and nutrient movement into the streams. The best available guidelines for buffer width associated with prescribed fire is that, low intensity fires that do not kill stream-shading shrubs and trees can be used throughout riparian areas without creating substantial damage. Where fire damages woody vegetation, the width should be proportional to the size of the contributing area, slope, cultural practices in the upslope area and the nature of the drainage below.

There are mixed concerns about leaving downed large woody debris in, or near, stream channels following fire. Large woody debris plays an important role in hydraulics, sediment routing, and channel morphology of streams flowing through riparian systems, thereby enhancing these systems. Sometimes, to protect life and property, debris is removed following wildfires. In general, large woody debris in streams after fire has numerous advantages and should be left on site unless downstream risks are readily identified.

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Fire Severity Effects on Water Resources

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Abstract.—Fire plays an important role in southwestern ecosystems. The use of fire, however, must be carefully planned and implemented to gain the desired response without damaging the water resources. Recognizing the differences between fire effects during prescribed burns and wildfires is necessary to adequately assess fire effects. This paper summarizes the basic hydrologic processes affected by fire, offers a conceptual model for relating watershed responses to fire severity, and presents generalized data for selected hydrologic responses of a ponderosa pine forest to fire in the Madrean Archipelago Province.

INTRODUCTION

Prescribed fire is an important tool for managing southwestern ecosystems. The use of fire, however, must be carefully planned and implemented to gain the desired response without damaging the water resources. It is important when assessing fire effects to clearly differentiate between prescribed burning and wildfires because the two differ widely. The magnitude of response resulting from the use of fire depends on several factors, the most important ones being the severity of the fire and the immediate post-fire precipitation regime. The effects of fire on a watershed can best be discussed by considering the effect that fire has on the individual processes in the hydrologic cycle. Although there is considerable information on fire impacts on watershed resources, most of these data have been collected after wildfires. Information available on the soil and water resource responses to lower severity prescribed fires is scant.

This paper summarizes the basic hydrologic processes affected by fire, offers a conceptual model for relating watershed responses to fire severity, and presents generalized data for selected hydrologic responses of a ponderosa pine forest to fire in the Madrean Archipelago. The conceptual model is used to illustrate the differences in responses between prescribed burning and wildfires for streamflow and sediment regimes. Although there are several important vegetation types in the Madrean Archipelago, ponderosa pine was selected to illustrate this model because it was the type where the greatest amount of hydrologic response data is available, particularly at different fire severities. Even for ponderosa pine the data base is limited and, therefore, only generalized responses could be developed. However, they do illustrate the large differences in watershed responses which can be expected between prescribed burns and wildfires.

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FIRE EFFECTS AND THE HYDROLOGIC CYCLE

The major effect that fire has on hydrologic responses is through the removal of plant and litter cover which protects the soil surface. The amount of plant and litter cover removed during a fire determines the magnitude of hydrologic responses that can be expected following fire if precipitation amounts and intensities are equal. A closer examination of the hydrologic cycle clearly identifies those processes most affected by fire. In general, the effect of fire on these responses can generally be classified into on-site and off-site effects (Baker 1990). Important on-site processes discussed are interception, infiltration, soil water storage, snow accumulation and melt, and runoff and surface erosion. The most important downstream processes are stream flow and water quality.

On-Site Processes.

Interception

The most obvious consequence of removing the plant material during a fire is its effect on precipitation interception. Interception is the process whereby vegetation, or litter on the soil surface, interrupts the fall of precipitation before it strikes the soil surface. Vegetative cover intercepts precipitation, dissipating energy that would otherwise strike a bare soil surface. Intermediate and low levels of cover often remain after low severity prescribed burns. Interception not provided by plant canopies is provided by surface organic litter. In situations where a canopy is destroyed, the persistence of prefire levels of surface litter can be important for soil surface protection. During severe fires most of the plant canopy and litter cover are completely destroyed and, as a consequence, little interception occurs. In contrast, when only small amounts of plant cover are destroyed by the fire during a low severity prescribed burn, most of the precipitation energy is dissipated by the remaining vegetation and litter.

Infiltration

Once precipitation reaches the soil surface it either infiltrates into the soil or runs off the surface. Infiltration is the amount of water that can move through the soil surface in a given time period. If more water is supplied than can infiltrate, the excess runs off rapidly as overland flow. The most important factors affecting infiltration and associated runoff are percentage of ground cover, vegetation cover type, soil texture and porosity, and amount of soil organic matter (Baker 1990). Another important soil property is the wettability of the soil. Water repellency formed as a result of fire and can cause rapid overland flow and erosion, particularly on steep slopes (DeBano 1981).

Soil Water Storage

Soil water storage is also affected by fire. The soil mantle is normally charged with water during periods of precipitation. Water infiltrates into the soil mantle and is stored until the soil reaches field capacity, at which time water begins draining from the soil and can eventually become streamflow. Much of the water stored in the soil mantle in lost by evapotranspiration during the growing season. In the Madrean Archipelago, most of the soil water is depleted by transpiration from vegetation during the prolonged dry periods in spring and fall. The amount of stored water that is lost depends upon the type and density of vegetation occupying the site. It has been estimated for the Colorado River Basin that about 95 percent of the precipitation arriving at the soil surface is lost by evapotranspiration, leaving only about 5 percent that can be realized as streamflow (Hibbert 1979). Generally grasses have shallow root systems and, therefore, are only able to use water from a few feet in the soil. In contrast, trees and shrubs have roots which can penetrate several meters into the soil and, thereby are able to extract water from throughout the soil mantle. The effect of fire on water storage results from the removal of vegetation, which lowers the evapotranspiration losses and increases the amount of water stored, particularly in the lower part of the soil mantle.

Snow Accumulation and Melt

Snow accumulation at the higher elevations could be affected by spotty fires, particularly if small openings are created in formerly dense tree canopies. After fire, not only does tree canopy not intercept snowfall, but additional snow is usually deposited directly in small openings in forest canopy because of increased turbulence (Ffolliott et al. 1989). The amount of opening created by a fire depends on the severity of the fire. Severe wildfires can destroy much of the tree canopy and thereby have little effect on snow accumulation. In contrast, when small openings are probably created by low severity fires some increased snow accumulation may occur. On all burned areas, charred organic material remaining after a fire can change the albedo of the soil surface which would induce earlier and more rapid snowmelt. Snow accumulation is only important at the higher elevation environments which are limited in area within the region.

Runoff and Surface Erosion

Erosion in the Madrean Archipelago area is typical of that which occurs in the arid southwestern United

States and can be viewed as an unsteady or discontinuous process which transports sediment from a source and through a channel system with intermittent periods of storage (Wolman 1977). This episodic process is more characteristic of dryland climates than of humid regions because the major cause of erosion in the Southwest is the "big" storm events. These big storms move material from various sources including material temporarily stored in the channel system. The disproportionate amount of sediment and debris moved during these major storms makes it difficult to define a "normal rate" of erosion either in the undisturbed forest stand or following a particular fire. However, the greater the amount of plant canopy and organic surface litter removed during a fire, the greater the chances for surface erosion during a given storm. Prescribed burns are designed so as to not completely consume excessive amounts of litter and vegetative cover and thereby offer better protection to the soil surface.

Off-site Effects

Stream Flow

There is little information available on the responses of stream flow to low severity prescribed fires in the Madrean Archipelago. However, because the major watershed responses to severe wildfires often last for only a couple years, and sometimes only for the first runoff season, we can assume that the detection of downstream effects caused by prescribed burning is going to be difficult, if not impossible (Baker 1990).

Water Quality

Water quality parameters include sediment and dissolved nutrients, the most important of which are nitrogen and phosphorus. Some of the soil nutrients may be adsorbed on the organic and inorganic sediment particles. Most increases in sediment and chemical components are short lived and water quality soon returns to pre-fire levels (Gottfried and DeBano 1990, Sims et al. 1981)

The nitrogen compound of most interest is nitrate-nitrogen because of its high solubility and lack of adsorption in the soil. However, high levels of nitrate-nitrogen are usually not produced directly by the fire as is the case with ammonium-nitrogen, which is strongly adsorbed by soil colloids and humus in the soil. Orthophosphate and organic phosphorus are usually the predominate phosphorus compounds appearing in streamflow.

CHARACTERIZING FIRE INTENSITY AND SEVERITY

It is important when discussing fire effects on soil and water to clearly differentiate between fire intensity and fire severity. Fire intensity is understood by fire behavior specialists to be the rate of energy release per unit of ground surface area and is proportional to flame height and rate of spread (Chandler et al. 1983). Because fire intensity measurements are difficult to relate to specific soil and water responses (Hungerford 1989), fire severity has been used to describe the amount of vegetation and soil changes associated with a particular fire (Wells et al. 1979).

Fire severity is a subjective classification is based on the appearance of the litter and soil after burning (Wells et al. 1979). In most forest and range prescribed burns and in many wildfires, fire is limited to the litter layer and other fine materials near the ground. In this case, both the vertical and horizonal dimensions of a fire are used to classify the severity of a burn. Three severity classes have been developed: light, moderate, and severe burns. Using this systems in forests, any particular spot in a fire is classified as being lightly burned if the litter and duff layers are scorched but not altered over the entire depth. Moderate burns char the litter and duff but do not visibly alter the underlaying mineral soil. All the organic layer is consumed and the mineral soil structure and color are visibly altered on severely burned spots. This criteria is then extended horizontally to classify larger areas or even an entire fire; this is done by determining the percentage of the total area that is severely, moderately, and lightly burned. An area is considered severely burned if more than 10 percent of the area has spots that are severely burned, more than 80 percent moderately or severely burned, and the rest lightly burned. In a moderately burned area, less than 10 percent of the area is severely burned but over 15 percent is moderately burned. A lightly burned area would have less than 2 percent severely burned, less than 15 percent moderately burned, and the rest lightly burned or non-burned.

The relationship between fire intensity and fire severity for different vegetation types remains largely unsolved, although progress is being made in developing quantitative models to describe changes in thermal conductivity in soils (Campbell et al. 1994), and soil temperature and water content beneath surface fires (Campbell et al. 1995). These relationships, once developed, will then be used to develop models that describe fire-driven heat and moisture transport in soils (Albini, In Preparation). However, because these quantitative models have not been fully implemented, the resource responses discussed in this paper will refer primarily to more qualitative descriptions of fire severity levels.

Because fire intensity basically deals with the amount and rate of surface fuel consumption, it may or may not be directly related to the amount of energy that is transmitted downward into the soil and the associated changes in the physical, chemical, and biological soil properties (Hartford and Frandsen 1992). It is possible to have a high intensity fire (such as a fast moving crown fire) which consumes little of the surface litter because only a small amount of the combustion energy is transferred downward to the litter surface. In this case, little damage of the surface organic litter can occur. However, if the crown fire also consumes substantial surface fuels and the residence time is greater, then a "white ash" layer might be the only material left on the soil surface.

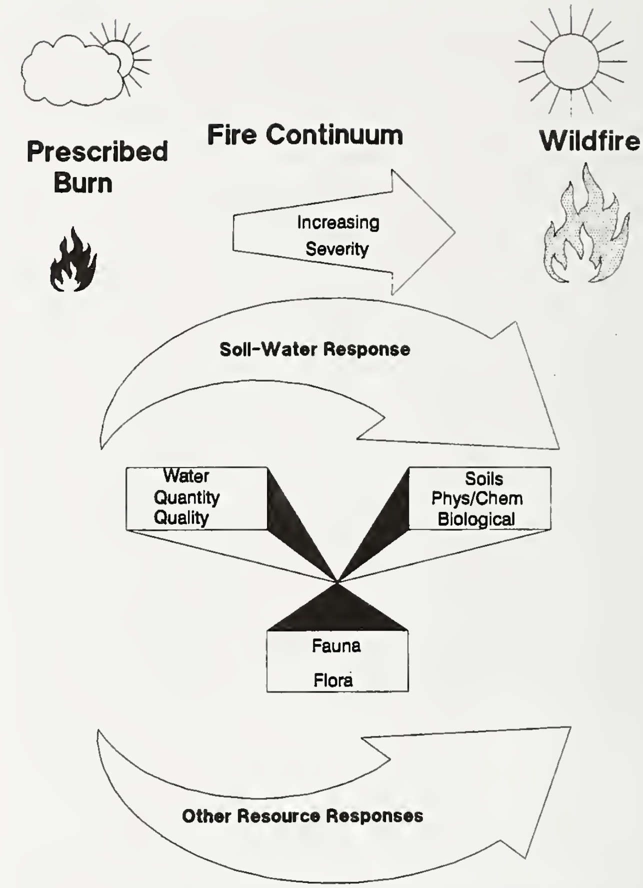

Figure 1. Conceptual model relating immediate resource responses to a fire severity continuum extending from cool-burning prescribed fires to severe wildfires.

CONCEPTUAL MODEL

A conceptual model is described that portrays fire severity as a continuum ranging from minor resource responses under a cool burning prescribed fire to major responses which could be expected to occur during stand-replacing wildfires in forests (DeBano et al. 1995). The fire response continuum in the Madrean ecosystems is large. In Figure 1, prescribed fire conditions are depicted on the left side of the fire response continuum and represent lower temperature-higher humidity burning conditions where fuel loading is minimal and fuel moisture is high. These conditions produce lower fire intensities and, thereby, expose the soil and water resource to lower fire severities. Prescribed fire usually has minor hydrologic impacts on watersheds because the surface vegetation, litter, and forest floor are only partially burned (Baker 1990, Sims et al. 1981). Other resources (soils, wildlife, vegetation, etc.) are also changed little by a prescribed fire. On the other end of the fire response continuum (right side of Figure 1), fire behavior more nearly represents that present during wildfires, where the temperatures, wind speeds, and fuel loadings are high, and humidities and fuel moisture are low. In contrast to prescribed burning, wildfires can have a major effect on basic hydrologic processes, leading to increased sensitivity of the site to eroding forces and to reduced land stability (Baker 1990). Large changes also occur in the other resources (denuded landscapes, large losses of plant nutrients, etc). Before this conceptual model can be implemented, the responses of different water resources to fire severities (extending from low to high severity fires) need to be defined.

The differences in impacts between prescribed burning and wildfires depend partly on the vegetation type being burned. For example, there can be large differences in ponderosa pine forests; during a cool prescribed fire, only the litter and smaller diam-

eter surface fuels are ignited as compared to near total canopy consumption during intense wildfires. In contrast, fires burning in brush dominated landscapes are mainly carried by the shrub canopies. Therefore, it is more difficult to control the behavior and intensity of the fire so that only minimum impacts to the soil and vegetation occur. To obtain less severe fires in brush-dominated areas, fires are ignited during marginal burning conditions or by using special heat generating ignition techniques (Helitorch, Ping Pong Balls, etc). Also, low severity fires in brush-grass areas often result in mosaics of burned and unburned patches because the slight differences in slope and aspect make total ignition and coverage impractical. In grasslands, both wild and prescribed fires are usually of low severity because the consumption of the light fuel loads does not produce significant soil heating and many of the underground grass parts are undamaged.

Another important factor affecting the postfire hydrologic scenario is the postfire precipitation pattern. The hydrologic response model is visualized as being a three dimensional surface, with a particular hydrologic response (peak flow, sedimentation rate, etc) being a function of both fire severity and a time variable reflecting climatic events following a fire. The immediate soil and watershed response would be most closely related to fire severity (how much litter and plant cover had been destroyed by the fire, amount of nitrogen volatilized, etc), and would probably be a non-linear function such as the one that describes infiltration into a wettable dry soil. An additional time function, reflecting precipitation events, would be necessary to define the longer-term hydrologic responses to a particular fire severity. The dimension of time is essential for the model because of the possibility of variable precipitation events that could follow a fire. For example, even a low intensity prescribed fire can produce substantial runoff and soil loss as sediment if the fire is immediately followed by high intensity rainstorm events. Conversely, severely burned watersheds can produce little runoff and erosion if a fire is followed by a relatively mild year with gentle rains and warm temperatures that allow a protective vegetation cover to develop. The pattern of precipitation events over time is stochastic in nature and must be viewed in a probability framework so that relevant outcomes can be explored. As noted earlier, the effects of "big" storms in the Southwest are of particular importance. It may be that low-probability, high impact events are more important in prescribed burning decision criteria than long-term average outcomes, reflected in the "expected values" of probability distributions. Information for the two-dimensional part of this model has been developed by Ffolliott et al. (1988), and with some modification could be used as a starting point for developing the time dimension following fire.

PONDEROSA PINE EXAMPLE

There is little information in the published literature which relates water resource responses to different fire intensities or severities. Some hydrologic response data are available for ponderosa pine forests (Campbell et al. 1977, Gottfried and DeBano 1990, Rich 1962, Sims et al. 1981, Zwolinski 1971). Likewise, some soil response data are also available for ponderosa pine (Covington and Sackett 1984, 1990; Wagle and Eakle 1979). Much less information exists for watershed and soil responses to fire in mixed conifer forest, encinal and pinyon-juniper woodlands, and desert grasslands, although there is pertinent information in nearby areas in the western United States that may be applicable to Arizona and the southwestern United States.

The first iteration of the fire response model introduced earlier is being developed to describe soil and watershed responses. All available information on hydrologic responses for southwestern United States is currently being consolidated to define and quantify hydrologic responses to both wildfires and, more importantly, cooler burning prescribed fires. Unfortunately, a meager data base is available on the hydrologic responses of watersheds to lower fire severities. The best data set available was collected during a case study of the Rattle Burn in north-central Arizona (Campbell et al.1977), and it was used to quantify the conceptual model described above for ponderosa pine forests.

A wildfire designated as the Rattle Burn (May 7-9, 1972) swept through 290 ha (717 ac) of even-aged stands of ponderosa pine growing on sedimentary-derived soils on the West Fork of Oak Creek of the Coconino National Forest at an elevation of 2,040m (6,700 feet) about 29 km (18 mi) southwest of Flagstaff. Three small watersheds, representing severe and moderate burns and an unburned control, were established to assess the effects of wildfire on hydrol-

ogy, soils, timber and forage production, and wildlife populations. On the moderately burned watershed, fire was generally confined to the forest floor. The intensity on the moderately burned site was calculated to be 9,000 kjoules/sec/m (2,500 BTU/sec/ft). On the severely burned watershed most of the trees were killed. The fire intensity on the severely burned watershed was calculated to be 35,000 kjoules/sec/m (10,000 BTU/sec/ft). A nearby unburned watershed served as a control.

Streamflow was measured between 1972 and 1975 with automatic stream stage recorders at the outflow points of each watershed. The mean annual water yields were 6 mm (0.2 in) from the unburned watershed; and 20 and 27 mm (0.8 and 1.1 in) from the moderately and severely burned watersheds, respectively. Although the differences in annual water yield were small between the watersheds, the differences in peak flows were much larger. The highest peak annual discharge on the control watershed was only 0.01 m3/s/km2 (0.92 ft3/sec/mi2). On the moderately burned watershed, peak discharge reached 0.24 m3/s/km2 (21.5 ft3/sec/mi2), while on the severely burned watershed the peak discharge exceeded 4.067 m3/s/km2 (355 ft3/sec/mi2). The number of runoff events during the period 1972-75 also increased with severity of burning, ranging from 6, 15, and 25 for the unburned, moderate, and severe burns, respectively. The increased peak runoff was also reflected in suspended sediment yields. The total suspended sediment yield for 1972-75 was 3, 18, and 1392 lb/ac for the unburned, moderate, and severe burns, respectively. The suspended sediment losses occurred during the first two years, and particularly during the second year when all time precipitation records were set.

The peak runoff and suspended sediment data from the Rattle Burn were used to establish maximum responses that might be expected from a ponderosa pine forest that had been subjected to different burning severities in situations similar to those on the Rattle Burn. Best estimates of the duration of annual streamflow responses were combined with the maximum numbers to generate the response curve for a particular fire severity (Figure 2 and 3). It is assumed, in general, that annual streamflow increases on watersheds burned at moderate severity will be less in magnitude and of much shorter duration than those for the severe burn. Probably after a couple growing seasons, little response will be evident on the moderately burned watersheds. In contrast, the response curves for watersheds receiving severe burns will be greater in magnitude and be longer in duration than for the moderately burned watersheds, perhaps lasting up to 15 years. Although no information was available on the low severity fire, it was assumed there would be no substantial increases in streamflow or sediment production.

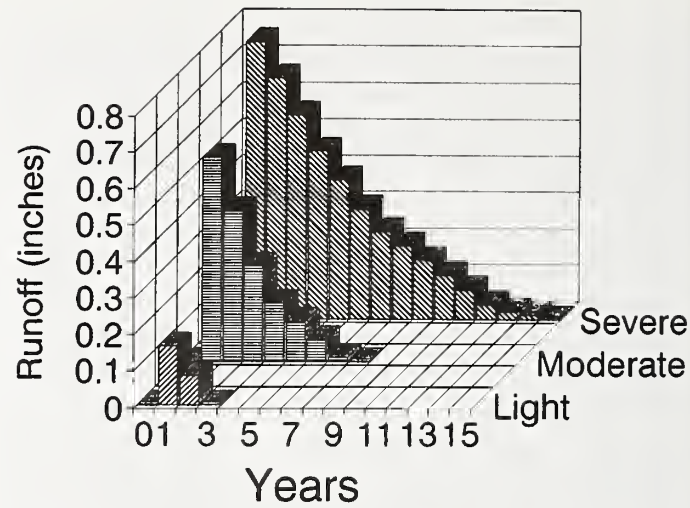

Figure 2. Average increases in annual runoff over time from ponderosa pine watersheds that were severely, moderately, and lightly burned.

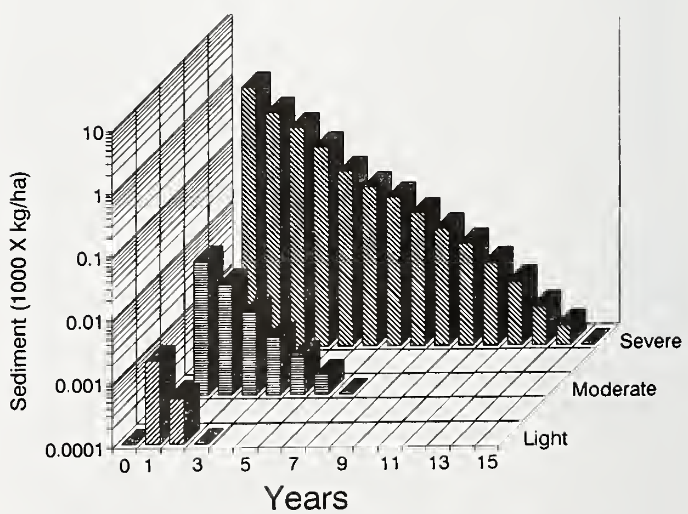

Figure 3. Average increases in annual sediment yield over time from ponderosa pine watersheds that were severely, moderately, and lightly burned.

DESERT GRASSLAND AND SHRUB

Recently, there has been a renewed interest in using prescribed fire in the management of desert grassland in southeastern Arizona. Over the last century many of the desert grasslands have been invaded by woody species, notably mesquite and creosotebush. One of the reasons given for the invasion of woody species is the exclusion of fire which formerly burned through these grasslands at regular intervals and suppressed invading woody species (Bahre 1985, McPherson 1995). As a result, many of the former grasslands are now dominated by dense covers of shrubs and other woody vegetation. The renewed use of prescribed fire has lead to the burning of areas which vary from almost pure stands of grass to areas dominated by woody vegetation; which results in fire severities which vary greatly, ranging from severe burns on brush occupied areas to low severity on the areas occupied primarily by grass.

The hydrologic response to burning shrub-invaded grasslands is not well known, although some general conclusions can be drawn from our experiences in other areas having similar vegetation. The brush-grass mixture would probably burn at widely differing severities over short distances, resulting in a distinct burned-unburned mosaic pattern. This burning pattern would leave some bare areas that are vulnerable to surface erosion. However, the effect of the severely burned sites on total runoff is most likely mitigated by interspersed areas of grass and woody vegetation which are either unburned or less severely burned. The end result is probably a minimum amount of surface erosion on gentle slopes that are not exposed to major storms immediately following the fire. The prescribed fire would affect soil water storage by reducing brush plants which are capable of depleting water to greater depths in the soil mantle. As a consequence, there would most likely be an increase in available moisture in the soil mantle for grass growth and streamflow. The streamflow effect would depend upon a number of factors including the nature of the mosaic (size and shape), distances from stream channels, soil characteristics, and aspect.

CONCLUDING STATEMENT

Although considerable information is available on hydrologic changes after wildfire, only limited information is available on such changes following prescribed fires. Because of the natural variability found in forest and range environments in the Madrean Archipelago region and in burning situations, fire influences can be viewed as a continuum, with effects of prescribed burning at one extreme and wildfire at the other. Low severity prescribed fires usually have minimal hydrologic impact on soil and water resources as the result of partial burning of the surface vegetation, litter and forest floor. Wildfires, on the other hand, can kill trees and other vegetation, and consume the forest floor over extensive areas on a watershed. As a result, wildfire can exert a pronounced effect on basic hydrologic processes, leading to increased sensitivity of the site to eroding forces and to reduced land stability. Fire generally increases overland flow, peak and total stream discharge, and sediment movement, but assessment of these increases must be related to fire severity.

Prescribed burning can be used as an economical tool in managing and manipulating vegetation in an ecologically sound manner in the Madrean Archipelago. Most land managers have responsibility for at least two or three vegetation types, and the influence of burning in these different ecosystems are usually different. However, these managers must be aware of the delicate balance that exists within the soil-plant-water-atmosphere system and how easily it can be influenced either positively or negatively by fire. Two major gaps in our knowledge are the cumulative effects of burning on watershed condition and the effects of burning in riparian habitats.

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Effects of Climate, Fire, Land-Use History, and Structural Development on Forest Communities

Jose Villanueva-Diaz and Guy R. McPherson[1]

Abstract.—Chronologies were developed for southwestern mountains with different land-use histories. Douglas-fir chronology surpassed other chronologies for climatic reconstructions. Seasonal precipitation (October-January) and current July PDSI were reconstructed for Animas Mountains, New Mexico. Annual precipitation (July-July) and current July PDSI were reconstructed for Sierra los Ajos, Sonora, Mexico. Fires in Animas Mountains were preceded by relatively wet conditions two years before the fire-year. No significant relationship was found between climatic variables and fire occurrence in Sierra los Ajos. Tree radial increase differed between mountain ranges suggesting that annual growth was influenced by differences in land-use history.

INTRODUCTION

Climate change is increasingly relevant and importance. Systematic meteorological observations are limited in extent or non-existent in the southwestern United States and northern Mexico. Extending of climate changes via proxies enables improved interpretation of the influence of climatic variables on the development of forest communities. Coupled with land-use history, this information may assist land managers by inferring the influence that climatic factors have had on forest dynamics.

Trees record climatic conditions that constrain physiological processes, and store that information in their annual rings (Fritts 1976). Therefore, tree-rings have been used as a source of proxy data in determining hydrologic and climatologic histories, ecological changes in forest communities, and modeling growth trends and events relating to forest decline and mortality.

Given the importance of understanding the effects of climatic factors on forest dynamics, the objectives of this study were to:

1. Use tree-ring series as proxy data for climatic reconstruction purposes;
2. Quantify relationships between climate and fire occurrence; and
3. Assess the role of land use on radial growth.

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STUDY AREAS

Animas and Sierra los Ajos Mountains

Geographical Location

The Animas Mountains (AM) are the highest range in southwestern New Mexico, west of the Rio Grande and south of the Mogollon Plateau (31° 35' N latitude, 108° 47' W longitude). The highest peak in the AM is 2600 meters. The AM extend over a 100 square-kilometer area in the Gray Ranch of southern Hidalgo County, NM (Fig. 1)

Sierra los Ajos (SLA) is located in Sonora, Mexico (30° 55' N latitude, 109° 55' W longitude), about 100 kilometers southwest of the AM. Its main peak rises above 2600 meters (COTECOCA 1973). The mountain range encompasses approximately 171 square kilometers.

Geology and Soils

The AM were formed by regional uplift and widespread volcanic activity during the Tertiary. During the Cretaceous, basalt rock was erupted followed by granodiorite stock. On the late Tertiary Period volcanic activity was renewed with eruptions consistent

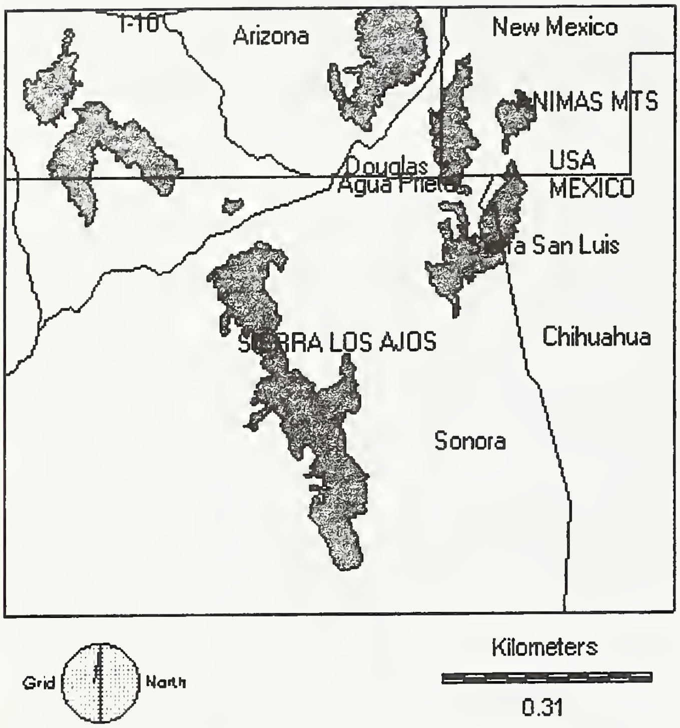

Figure 1. Geographical Location of Animas Mountains, New Mexico and Sierra los Ajos, Sonora

of rhyolite, tuffs, welded tuffs, and basalt (Arras 1979, Stone and O'Brian 1990, Wagner 1977).

The SLA has a complex geologic formation characterized by an heterogeneous lithic composition (Aponte 1974). Outcrop from Precambrian and Holocene age are found along the altitudinal gradient.

The AM and the SLA are characterized as having a rough topography with extreme slopes. Rocky to shallow soils covered with cobbles and gravel are characteristics of these mountains, with soil depths ranging from 10 to 50 centimeters (Soil Conservation Service 1973, Garza-Salazar 1993).

Climate

The AM and the SLA have a bimodal precipitation pattern with about 60 percent of the average annual precipitation (450 - 750 millimeters, depending on elevation) falling in July-September and 40 percent received in the winter months. Temperatures above 32°C are common during the summer months and usually range between 12°C to -5°C during the winter months. An average evaporation of 2340 millimeters per year can be expected for both mountain ranges (Tonne at al. 1992, Solis-Garza et al. 1993).

Vegetation

The proximity of both the AM and the SLA to the Sierra Madre and the southern Rockies has resulted in a floristically diverse vegetation composition comprised of both northern and southern elements (Wagner 1977, Brown 1982, Fishbein et al. 1995).

Wagner (1977) categorized the vegetation of the AM (main massif) into three basic types: lower encinal, upper encinal, and forests. Forest community covers approximately 450 hectares between 1980 an 2600 meters elevation (Hubbard 1977). Major species present in this vegetation type are Douglas-fir (*Pseudotsuga menziesii*), found in canyons and on ridges of the northern portion of the mountain. Scattered within this species are southwestern white pine (*Pinus strobiformis*), ponderosa pine (*Pinus ponderosa* var. *Arizonica*), and chihuahuan pine (*Pinus leiophyla* var. *chihuahuana*).

The mixed pine forest type is found between 2300 and 2600 meters elevation and is composed of ponderosa pine, southwestern white pine, chihuahuan pine, and apache pine (*Pinus engelmannii*) (Wagner 1977).

The pinyon pine-juniper woodland community is found at elevations between 2300 and 2450 meters, and is composed by pinyon pine (*Pinus discolor*), alligator juniper (*Juniperus deppeana*), and oaks (*Quercus spp*).

The SLA supports biotic communities classified as mixed conifer forest, montane meadows, montane chaparral, oak woodland, and riparian forest (Garza-Salazar 1993, Fishbein et al. 1995). The mixed conifer forest is restricted to north-facing slopes between 1900 to 2600 meters elevation. North aspects are dominated by Douglas-fir, associated with gambel oak (*Quercus gambelii*) and madroño (*Arbutus arizonica*).

The pine-oak association is found between 1500 to 2000 meters elevation, associated species are ponderosa pine, chihuahuan pine, pinyon pine, emory oak (*Quercus emoryi*), silverleaf oak (*Quercus hypoleucoides*), and Arizona oak (*Quercus arizonica*).

HISTORICAL LAND USES

Animas Mountains

Knowledge of land-use history and human-related disturbances is important to understanding

and interpreting changes in forest composition and age structure. Human occupancy of southwestern New Mexico and southeastern Arizona has been recorded for more than 10,000 years (Martin 1963). Spanish settlers moved into the region in the early 1700. Their primary activities were farming and cattle raising, but they also began to mine some of the region's mineral resources. Grazing intensities in the AM during these early days are not reliably known (Wagner 1977). Large-scale grazing began during the 1890s, when it became one of the three ranches comprising the historic Diamond A Cattle Ranch (Tonne et al. 1992). Although most of the grazing activity took place in grasslands, remains of water tanks, forest trails, and old fences indicate that forest communities also were affected.

Fire is natural component of forest ecosystems that has been particularly affected during this century. Grazing and post-1900 fire suppression in the southwestern Unites States may have triggered several environmental and vegetation changes, including arroyo cutting, invasion of desert grasslands by shrubs, and replacement of shade tolerant species by shade intolerant species (Archer and Smeins 1991).

The fire regime for the AM before 1900s is characterized by a mixture of low-intensity, short-interval surface fires (3-15 years), and higher-intensity fires at longer intervals (20-50 years). Post-1900 fires for the AM have been reported by Baisan and Swetnam (1995).

Sierra los Ajos

Land-use history of SLA is relatively unknown. The region was inhabited by Opata tribes (Hasting and Turner 1965, West 1993). Spanish settlement of the region did not occur for nearly a century after the area was searched for mine: from 1614 to 1617, the Jesuit order began to establish missions in Sonora. Rising livestock became an occupation and way of life for many settlers in eastern Sonora. The Jesuit economic system included the raising of livestock on a comparatively large scale. However, a increase in livestock and consequent impact on herbaceous vegetation was produced by the establishment of cattle ranches in the early 1700s. For example, one rancher owned 7,000 head of cattle in the Moctezuma valley by 1713; at the same time, six Spaniards were running at least 12,000 head of cattle in the mountain between the Sonora and the Moctezume valleys (Harness and Barber 1964).

Current livestock grazing activities in SLA are not well documented. Several grazing permits have been reported for the period 1968-1984, but the intensity of grazing and number of animal units is unknown. Current human use of this mountain centers on cattle ranching, especially at lower elevations.

The occurrence of fire in the SLA has not been reduced by suppression activities. Fire frequency for the SLA varies from 4 to 5 years (Dieterich 1983, Baisan and Swetnam, 1995).

METHODS

During 1992 and 1993 permanent plots were established in each of four representative stands of three forest communities in both mountain ranges: (1) Douglas-fir/gambel oak forest, (2) mixed-pine forest, and pinyon pine/juniper/oak woodland. One 20 X 50 m (0.1 hectare) permanent plot was established in each stand. Each plot was divided into ten 10 X 10 meter subplots in which two sound increment cores were collected from each of two randomly-selected living trees (n= 20 trees/plot). Increment cores were prepared and dated following standard dendrochronological techniques (Stokes and Smiley 1968, Swetnam et al. 1985).

To assign each tree ring its exact year of formation, skeleton plots for Douglas-fir, southwestern white pine, ponderosa pine, and pinyon pine were constructed for each core and crossdated with composite skeleton plots. Crossdating was verified and corrected with COFECHA (Holmes 1994).

The program ARSTAN (Cook 1985, Holmes 1994) was used to detrend each ring series. Linear or negative exponential single detrending was chosen to find the best fit for each series. Ring-width chronologies were developed for individual species and for all species combined at each mountain range.

Descriptive statistics (mean sensitivity, standard deviation, first-order autocorrelation, and signal-to-noise ratio) were compared between and within mountain range chronologies. A chronology is well-suited for dendroclimatological applications if the following conditions are met: high frequency variations, large standard deviation, low first-order autocorrelation, and high values of cross correlation (Stockton et al. 1985, Fritts 1991).

Within-and between-mountain range chronologies were compared to define the response of trees to

environmental conditions. A t-test analysis was used to define the statistical significance of that response.

Dendroclimatic Analysis

Relationships between tree-ring index chronologies and climate were investigated using response function and correlation analyses (Fritts 1976, Guiot et al. 1982). To define those months in which temperature and precipitation were correlated with tree growth, weather stations near to the AM and to the SLA were considered, as well as regional climatic data (precipitation, temperature, and PDSI values) developed for NOAA Climatic Divisions over the period 1896-1993. Monthly total precipitation and monthly average temperature were seasonalized for the previous and current year seasons, since the tree integrates the effects of favorable or poor climatic conditions over more than one growing season (Fritts 1976).

PDSI, which is a meteorological drought index based on monthly computations of moisture supply and demand (Palmer 1965, Karl 1986) was correlated with ring-width index .

Calibration and Verification

Calibration involves the development of a mathematical or statistical transfer function which is used to extract the climatic signal from tree rings (Cook 1992, Fritts 1976, Lofgren and Hunt 1982, Stockton et al. 1985). Calibration equations were developed to predict climate (precipitation and temperature) and PDSI from a chronology for sub-periods (1896-1930 and 1931-1993) using least-squares analysis. calibrated equations were verified by testing performance on the sub-period withheld during calibration. The roles of the two sub-periods were then reversed, and the model was calibrated opposite. Predicted climate values were compared to actual climate values using correlation analyses, reduction of error tests, nonparametric sign tests, and the product mean test (Conkey 1979, Fritts, 1976, Gordon 1982).

If the calibration-verification procedure for both subperiods proved to be statistically significant for climatic reconstruction, then the whole period (1896-1993) was considered to develop a new linear regression equation to reconstruct the selected climatic variable for the entire length of tree-ring chronology. Once reconstruction was completed, a 10-year smoothing spline was produced for reconstructed precipitation and reconstructed PDSI for each mountain range.

Decadal-length climate episodes may produce information related to dry or wet episodes that then may be related to stand dynamics (Fritts and Swetnam 1989). Dry and wet episodes were characterized by determining the long-term mean and confidence intervals for precipitation and PDSI of the entire reconstruction.

Superposed Epoch Analysis (SEA)

Fire is a natural disturbance that has important implications for vegetation dynamics. Several studies have indicated the strong climate-fire relationship in the southwestern United States (Baisan and Swetnam 1990, Grissino-Mayer 1995, Touchan and Swetnam 1995). To analyze the influence of mean climatic events before and after fires a SEA (Baisan and Swetnam 1990) was considered. In this analysis, SEA was used to compare fire dates with precipitation and PDSI reconstructed from Douglas-fir in the AM, and precipitation and PDSI reconstructed from ponderosa pine in the SLA. SEA was used to describe short-term (<10 years) relationships between fire and tree growth.

Tree Growth: Radial Increase

To describe relationship between land-use history and tree growth, average ring-width series were generated for Douglas-fir, ponderosa pine, southwestern white pine, and pinyon pine in each mountain range. Annual average ring-width values for similar species between mountain ranges were compared with t-test.

RESULTS AND DISCUSSION

Quality of the Tree-Ring Data

The Douglas-fir chronologies for the AM and the SLA surpassed those of other species for climatic reconstruction as evidenced by lower autocorrelation, higher values for mean sensitivity, standard deviation, signal-to-noise ratio, and variance in the first eigenvector (Table 1). AM chronologies were generally superior to SLA chronologies for climatic reconstruction. This pattern is consistent with studies that indicated the most suitable chronologies for dendroclimatic inferences in western North America are located between 26° and 49° north latitude (Stockton

Table 1. Descriptive statistics for chronologies produced from tree species in Animas Mountains, New Mexico and Sierra los Ajos, Sonora

Statistic	Df	SWP	Py	Species* Comb.	Df	PP	Py	Comb.
	Animas Mountains				Sierra los Ajos			
Mean	1	1	1	1	1	1	1	1
Mean sensitivity	0.43	0.35	0.25	0.38	0.30	0.25	0.24	0.23
Standard deviation	0.40	0.40	0.25	.36	0.31	0.27	.23	.26
Skewness	0.36	0.66	0.61	.22	0.35	0.02	-.05	.54
Kurtosis	0.49	1.91	4.25	.39	2.32	0.76	.58	6.64
First-order autocorrelation	0.20	0.44	.14	.21	0.30	0.37	.26	.27
Second-order autocorrelation	0.02	0.20	.12	.02	0.03	-0.08	.17	0.10
Third-order autocorrelation	0.18	0.04	-0.02	.18	-.18	-0.06	-.04	-.15
Variance due to regression %	6.90	14.90	--	--	9.70	14.20	11.0	10.10
Error variance	0.01	0.02	--	--	0.03	0.01	.02	.01
Corr. among all radii	0.52	0.42	.34	0.41	0.40	0.38	.31	.27
Corr. between trees	0.52	0.40	.32	0.41	0.37	0.33	.29	.26
Corr. within trees	0.70	0.70	.55	0.70	0.56	0.77	.49	.57
Signal-to-noise ratio	22.64	4.57	6.86	30.04	4.78	2.46	3.32	8.22
Agreem. with population chrn.	0.96	0.82	.87	0.97	0.99	0.72	.77	.89
Variance in first eigenvec. (%)	55.70	48.70	38.30	44.40	43.70	48.10	37.30	31.60
Chron. common interval mean	0.96	0.97	1.00	0.99	0.99	0.86	.98	.99
Chron. common interval SD	0.36	0.30	.24	0.29	0.22	0.24	.18	.18

*Df, *Douglas fir;* SWP, *Southwestern white pine;* PP, *Ponderosa pine;* Py, *Pinyon pine;* Comb, *All speices*

et al. 1985). Trees growing on lower latitudes tend to be less constrained by climatic factors (Schulman 1941).

Within AM for 1860-1992 correlation between species exceeded 0.5. Correlations between species tended to be lower for chronologies within SLA. Correlations between chronologies were especially low before 1900, presumably because of relatively low sample sizes.

Ring-width index of all species within a mountain range were plotted to analyze their annual behavior (Fig. 2). In general, species within a given mountain range responded similarly to extreme climatic events. These results could be expected taken into account the similarity in response of a species to environmental factors.

Statistical comparisons within and between mountain range chronologies were not significantly different (P> 0.05) for any one of the species considered (Table 2).

Tree-Ring Chronologies and Their Association to Climate

The most significant associations between climate and tree growth in both the AM and the SLA were found using divisional climate data rather than using climatic data from single-weather stations. In the AM seasonalized climatic data, deduced from response function and correlation analysis, were particularly correlated with ring-width index (Table 3). Precipitation and PDSI were highly-correlated with tree growth; in contrast, ring-width index explained only 24-29 percent of the variance in average temperature, which is considered exceedingly low for reconstruction purposes. July PDSI was used for climate reconstructions in both mountain ranges.

For SLA divisional regional climate data for Arizona NOAA Climate Division 07, produced a more significant relationship between climate and tree growth than using single-weather stations.

Seasonalized previous July to current July precipitation indicated that 36 percent of the variance (r = 0.60) in seasonal precipitation data could be explained by the tree-ring data (Table 3).

Seasonal average temperature from the period March to July indicated a significant negative correlation coefficient (r =-0.53), which represents 28.5 percent of the variance explained by the tree-ring growth. Reconstruction of temperature was not produced due to the low variance explained by tree growth. PDSI was significantly correlated with tree growth from the period March to July. Maximum correlation coefficient of 0.67 was obtained for the

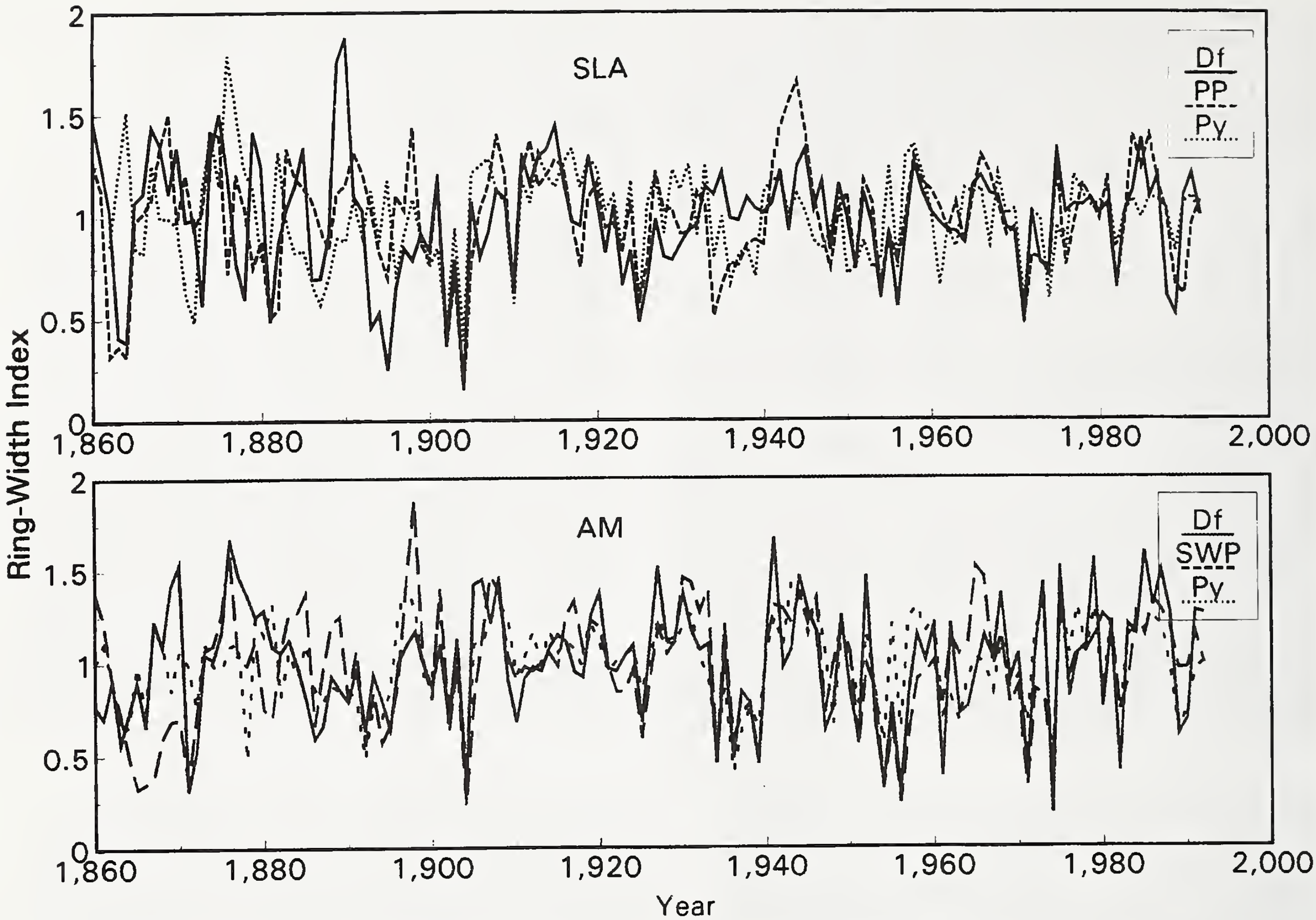

Figure 2. Chronologies produced within a mountain range emphasizing annual ring-width differences

Table 2. Comparisons of ring-width index values for chronologies within (a), and between mountain range (b)

Species*	N	r	Mean	St. Dev.	SEMean	t	P	Signif.
				a. Animas Mountains				
Df -SWP	132	0.594	0.0059	0.2919	0.0254	0.23	0.82	NS
Df - Py	132	0.597	-0.0115	0.2664	0.0232	-0.50	0.62	NS
SWP - Py	132	0.53	-0.0524	0.2895	0.0252	-0.69	0.49	NS
				a. Sierra los Ajos				
Df - PP	130	0.56	-0.0148	0.2706	0.0237	-0.62	0.53	NS
Df - Py	130	0.23	-0.0243	0.3304	0.029	-0.84	0.40	NS
PP - Py	130	0.32	-0.0095	0.3016	0.0265	-0.36	0.72	NS
				b. Animas Mountains - Sierra los Ajos				
Df - Df	156	0.401	0.0059	0.2919	.02	-.62	.53	NS
SWP - PP	133	0.501	-0.0115	0.2664	.03	-.84	.40	NS
Py - Py	130	0.458	-0.0524	0.2895	0.029	-0.36	0.4	NS

NS, *Non-significant at* P<0.01

* *Douglas-fir;* SWP, *Southwestern white pine;* PP, *Ponderosa pine;* Py, *Pinyon pine*

Table 3. Highest correlations between mountain range chronologies and divisional climatic data from Arizona and New Mexico*

Season or Month	r	Season or Month	r
Animas Mountains		**Sierra los Ajos**	
Previous Oct - Current Jan PPT	0.62	Previous Jul - Current Jul PPT	0.6
Previous Oct - Current Feb PPT	0.6	Previous May -Current Jul PPT	0.59
Previous Nov - Current Mar PPT	0.57	Previous Jun - Current Jul PPT	0.59
Previous Nov - Current Feb PPT	0.54	Previous Jul - Current Jun PPT	0.51
Previous Dec - Current Apr PPT	0.53	Previous Oct - Current Jul PPT	0.51
Previous Dec - Current Mar PPT	0.52	Previous May - Current Jun PPT	0.51
Previous Nov - Current Jan PPT	0.5	Previous Jun - Current Jun PPT	0.51
Current Mar - Current Jul TMP	0.49	Current Mar -Current Jul TMP	0.53
Current Jul PDSI	0.64	Current Jul PDSI	0.67
Current Jun PDSI	0.61	Current Apr PDSI	0.57
Current April PDSI	0.60	Current Mar PDSI	0.56
Current May PDSI	0.59	Current May PDSI	0.53

**All r values significant at P<0.01*

month of July, representing 45.2 percent of the variance explained by tree-ring growth. Therefore, PDSI for July was used for reconstruction purposes.

Correlations between precipitation and growth rate in these mountain ranges were low relative to other chronologies in the southwestern region (e.g., El Malpais Long Chronology, r = 0.76; Jemez Mountains chronology, r = 0.79) Grissino-Mayer 1995, Touchan and Swetnam 1995. Relatively low correlations may have resulted from the absence of reliable nearby weather stations (forcing us to rely on regional climatic data) or non-climatic factors such as variable land-uses (e.g., fire suppression, timber harvesting, livestock grazing).

Association Between Chronologies and Climate for Calibration and Verification

Final calibration models were significant ($p < 0.05$) for precipitation, temperature, and PDSI (Table 4).

Table 4. Calibrations between climatic variables and tree growth for Animas Mountains, New Mexico and Sierra los Ajos, Sonora*

Mountain range	Variable	Period	Parameter estimate	Constant	F	R-sq (Adj.)
AM	PPT	1896-1930	2.05	-0.13	41.2	0.598
AM	PPT	1895-1992	2.26	0.417	57.44	0.391
AM	TMP	1931-1992	-1.63	67.4	10.78	0.229
AM	TMP	1895-1992	-1.65	67.4	28.63	0.231
AM	PDSI	1931-1992	6.71	-6.21	143.7	0.722
AM	PDSI	1896-1992	5.41	-5.6	89.69	0.491
SLA	PPT	1896-1930	7.09	9.81	52.91	0.619
SLA	PPT	1896-1992	11.0	6.37	73.72	0.441
SLA	TMP	1895-1930	-2.07	63.5	29.27	0.454
SLA	TMP	1895-1993	-1.70	63.1	38.45	0.285
SLA	PDSI	1931-1993	5.69	-5.78	56.77	0.508
SLA	PDSI	1896-1992	4.53	-4.92	62.25	0.416

** All values significant at P<0.05*

PPT, *precipitation;* TMP, *temperature;* PDSI, *Palmer Drought Severity Index*

AM, *Animas Mountains;* SLA, *Sierra los Ajos*

Verification tests indicated that precipitation and PDSI reliably predicted tree growth in both mountain ranges (Table 5). Temperature was less reliably, thus casting further doubt on its use for climatic reconstructions.

The ability of precipitation and PDSI to predict tree growth as indicated by calibration and verification tests, suggested that a new calibration model could be developed for the entire period containing climatic data (i.e., 1896-1993). We used these calibration models to estimate PDSI and precipitation in the AM for the period 1760-1992, and PDSI and precipitation for the SLA for the period 1838-1992. Comparisons between actual and reconstructed data confirmed that the calibration model adequately simulated high frequency variability for PDSI and precipitation in the AM and the SLA (Fig. 3).

Climatic Trend Analysis

The AM region has experienced short-term (<10 years) substantial fluctuations in precipitation during the last 250 years (Fig. 4).

Above-normal precipitation for the AM occurred during the periods 1783-1788, 1790-1800, 1849-1852, 1876-1880, 1905-1908, 1918-1933, and 1975-1992. Since 1900, above-normal precipitation has been related to El Niño episodes (i.e., 1915, 1919, 1926; 1958, 1983 and 1993), or decades of frequent El Niño events and expanded circumpolar vortex (1900-1930 and 1960-1993). The latter event is characterized by greater precipitation in fall, winter, and spring, with slightly reduced rainfall during the summer (Andrade and Sellers 1988; Betancourt et al. 1993). Below-normal precipitation occurs during La Niña events (i.e, 1904,

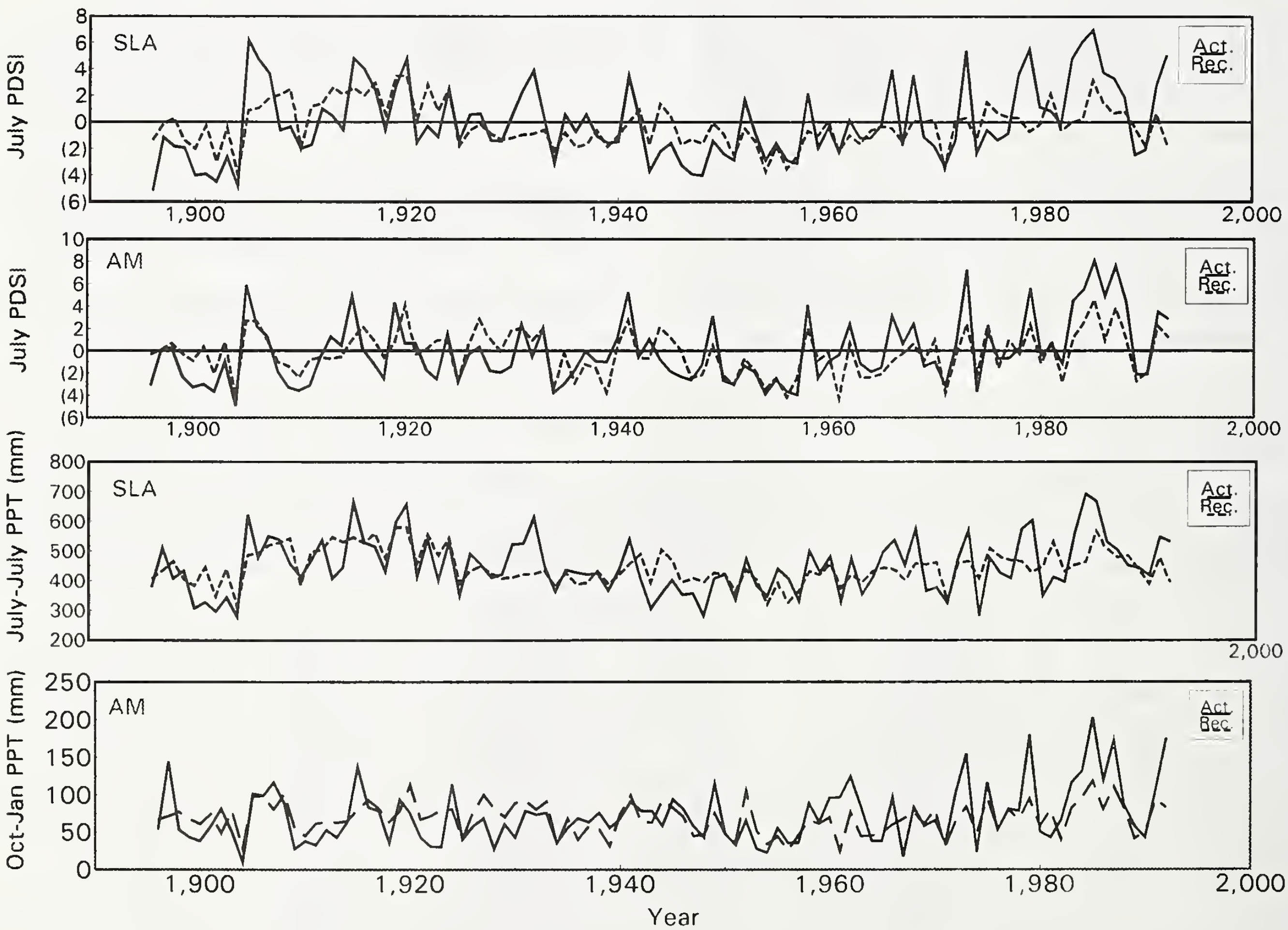

Figure 3. Comparisons between actual and reconstructed climatic data (precipitation and PDSI) for Animas Mountains, New Mexico and Sierra los Ajos, Sonora

Table 5. Results of verification tests between actual and predicted climate in Animas Mountains, New Mexico and Sierra los Ajos, Sonora*

Variable	Calibration period	Verification period	r	Reduction of error	Sign test	t
			Animas Mountains			
PPT	1896-1930	1931-1992	0.49	0.15	9	1.16
PPT	1931-1992	1896-1930	0.68	0.46	15	0.46
TMP	1895-1930	1931-1992	-0.44	0.17	14 NS	2.87
TMP	1931-1992	1895-1930	-0.29	-0.46 NS	23	3.56
PDSI	1931-1992	1896-1930	0.75	0.53	11	3.76
PDSI	1895-1930	1931-1992	0.48	0.15	12	3.85
			Sierra los Ajos			
PPT	1896-1930	1931-1993	0.74	0.55	8	3.37
PPT	1931-1993	1896-1930	0.51	0.25	21	4.04
TMP	1895-1930	1931-1992	-0.62	0.33	7	4.46
TMP	1931-1992	1895-1930	-0.27	0.04 NS	26 NS	2.45
PDSI	1931-1993	1895-1930	.57	.29	20	4.46
PDSI	1895-1930	1931-1993	.71	.49	9	4.67

* *All values are significant at P<0.05 except as noted by* NS

PPT, *precipitation;* TMP, *temperature;* PDSI, *Palmer Drought Serverity Index*

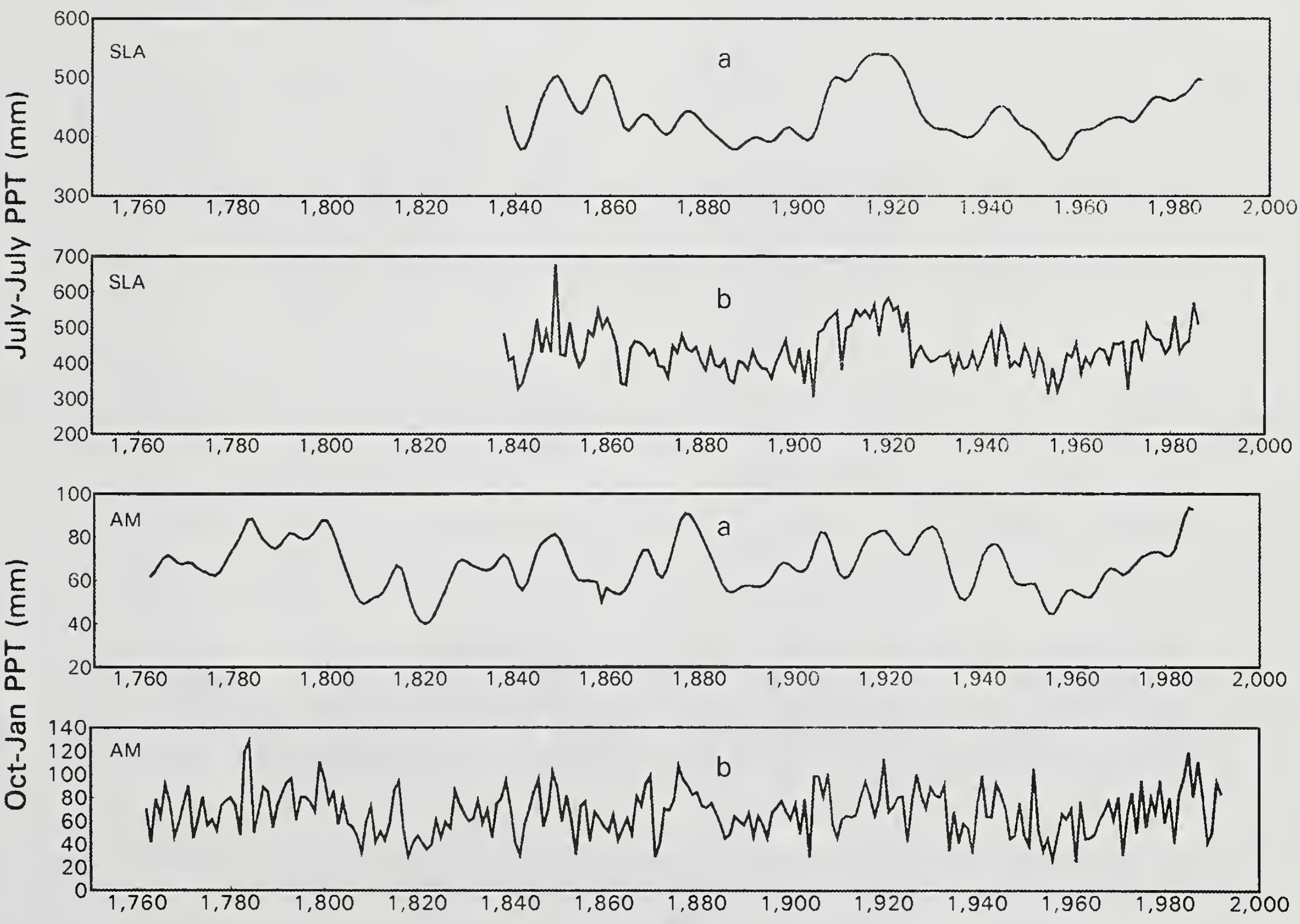

Figure 4. Reconstructed precipitation for mountain ranges emphasizing short-term decadal-scale precipitation differences (a) and annual precipitation fluctuations (b)

1917, 1925, 1943, 1950, 1955, 1974, and 1989) or decades when the circumpolar vortex contracts (Betancourt et al. 1993). Short-term periods of below-normal precipitation occurred during the periods 1807-1813, 1817-1825, 1899-1904, 1934-1939, and 1947-1957. All of these periods have been reported in other southwestern precipitation reconstructions (D'Arrigo and Jacoby 1992, Fritts 1991, Grissino-Mayer 1995, Touchan and Swetnam 1995).

Reconstruction of annual precipitation in the SLA for the last 155 years (1838 - 1993) indicated the presence of dry and wet episodes, coinciding with similar periods in the AM and other southwestern climatic reconstructions (Fig. 4).

Reconstructed July PDSI indicated a similar trend for both mountain ranges. In general, reconstructed PDSI detected wet and drought spells matching with similar periods recorded by the reconstructed precipitation (Fig. 5).

Because we relied solely on living trees to develop our chronology, a study designed to increase sample depth by using dead woody material may extend back climatic reconstructions in time. Considering, the high correspondence between chronologies from disparate sites, the great length of some existing chronologies, and the limited utility of climatic reconstructions for current and future resource management, initiating additional research in this area is not strongly recommended.

Superposed Epoch Analysis

Two years before a fire, tree growth was significantly higher than tree growth during fire years in

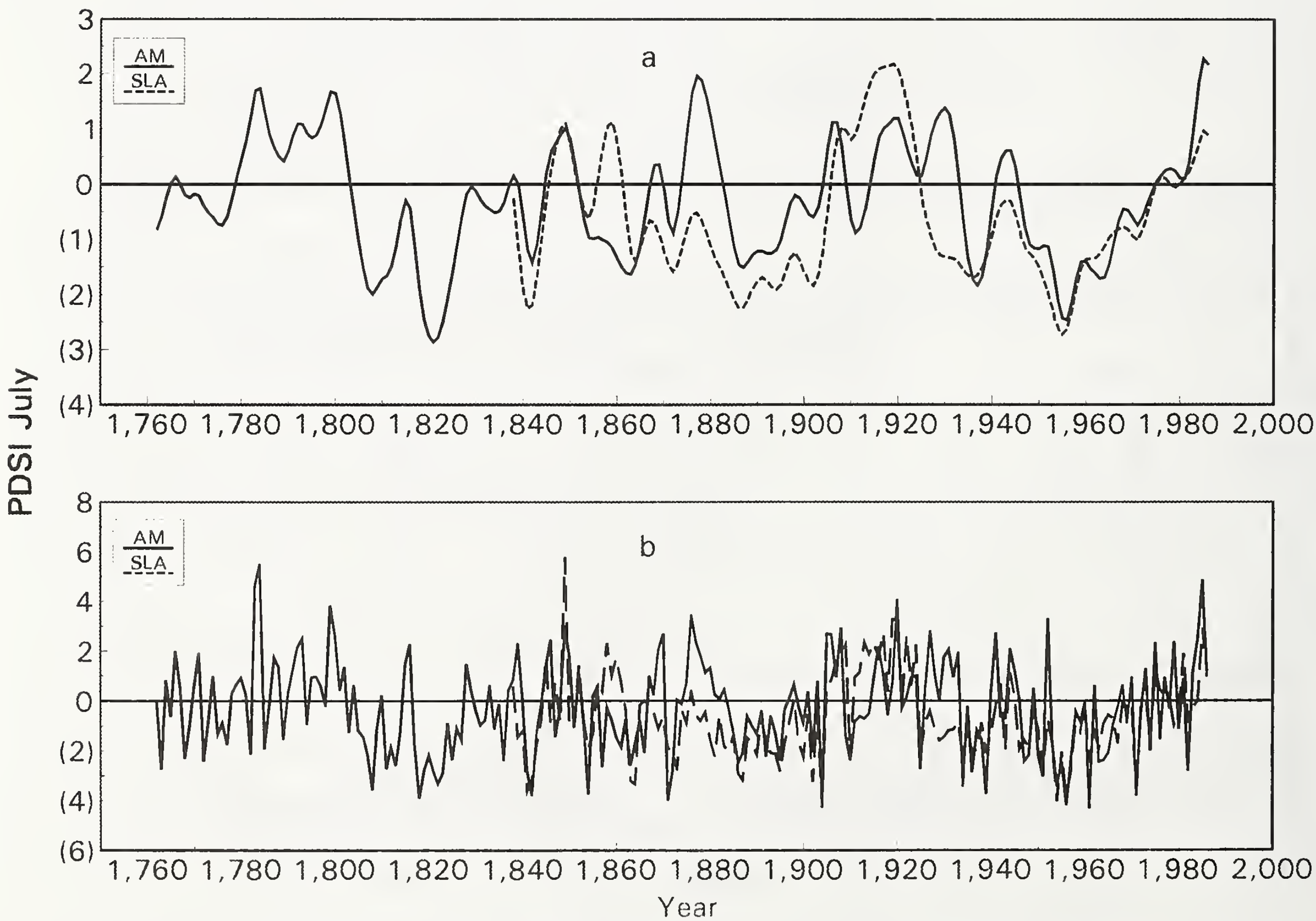

Figure 5. Reconstructed PDSI for mountain ranges emphasizing short-term decadal-scale PDSI differences (a) and annual PDSI fluctuations

the AM (Fig. 6). and Rogers and Vint (1987) discussed a similar fire-climate interaction for some areas of the Sonoran Desert. They found an increased probability of fire occurrence following two years of above-average precipitation. Higher probability of fire was linked to increased production of herbaceous winter annuals in the Sonoran Desert, and apparently also in nearly mountains (e.g., Baisan and Swetnam 1990).

PDSI also was correlated with fire occurrence. Mean PDSI was significantly positive for two years preceding fires indicating above-average moisture conditions. PDSI during the fire year was significantly negative, indicating that fire occurred during particular dry years.

Drought-fire relations for forest communities in SLA were not significant (Fig. 6). The absence of a pattern in the SLA may be attributed to the lack of extensive fire-history information, relatively frequent fires through the current time (e.g., Fulé and Covington 1995, Villanueva and McPherson 1995) or presence of other recurrent disturbances (e.g., timber harvesting, livestock grazing, smelter emissions) that mask fire climate relationships. Average Ring-Width Growth

Growth of pinyon pine differed between mountain ranges throughout most of the period of comparison (Fig. 7). Growth rates were lower in the AM relative to the SLA. Differences in tree growth may be attributed to local environmental conditions as well as to differences in land-use history. Pinyon pine communities in the AM experienced substantially lower fire frequencies than those in the SLA especially after the 1870 (Bahre 1991). This situation may have favored an increase in seedling establishment,

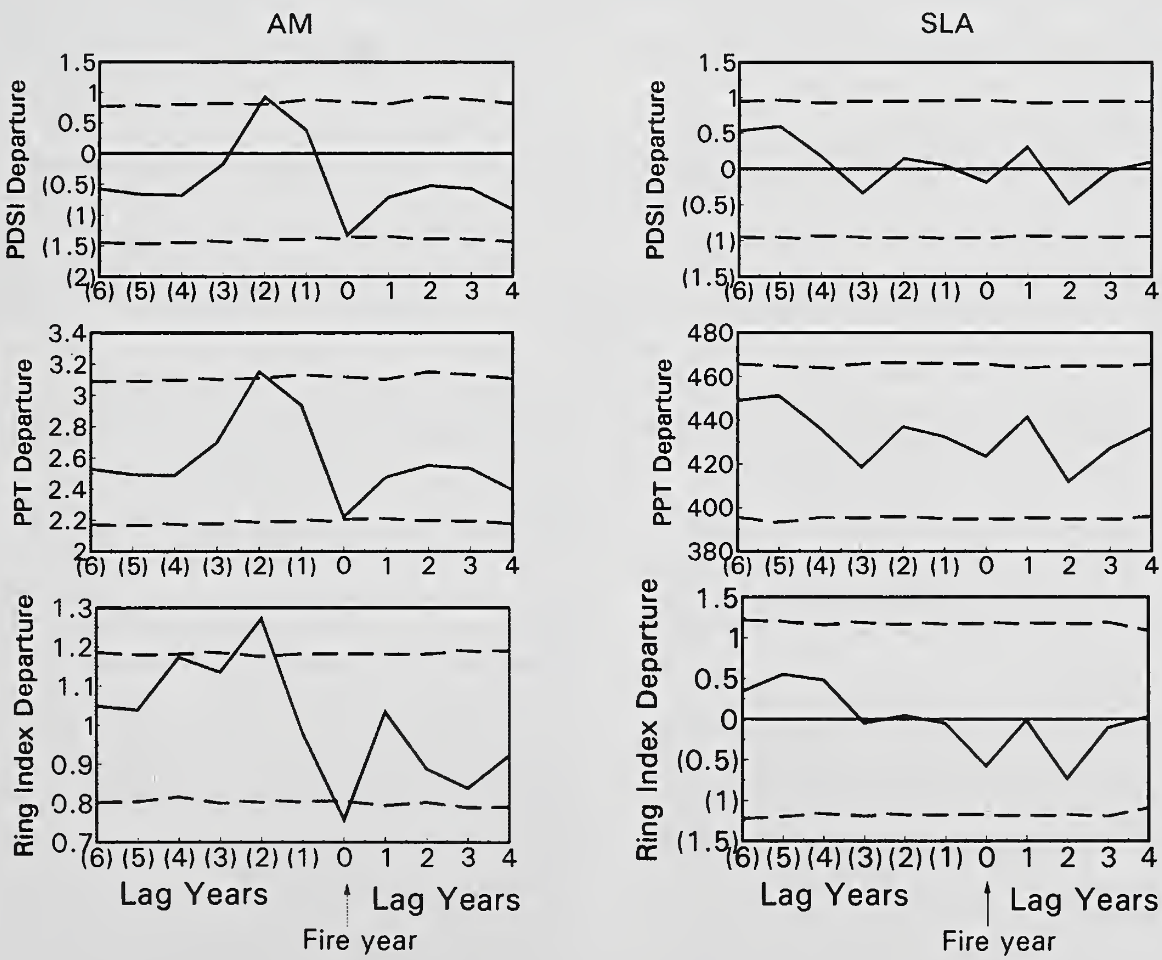

Figure 6. Superposed-Epoch Analysis for mountain ranges. Departures for PDSI, PPT, and ring-width index were computed as the difference between the long-term mean for those variables and their respective observed values during fire years and lagged years. Dashed lines represent 95% confidence intervals

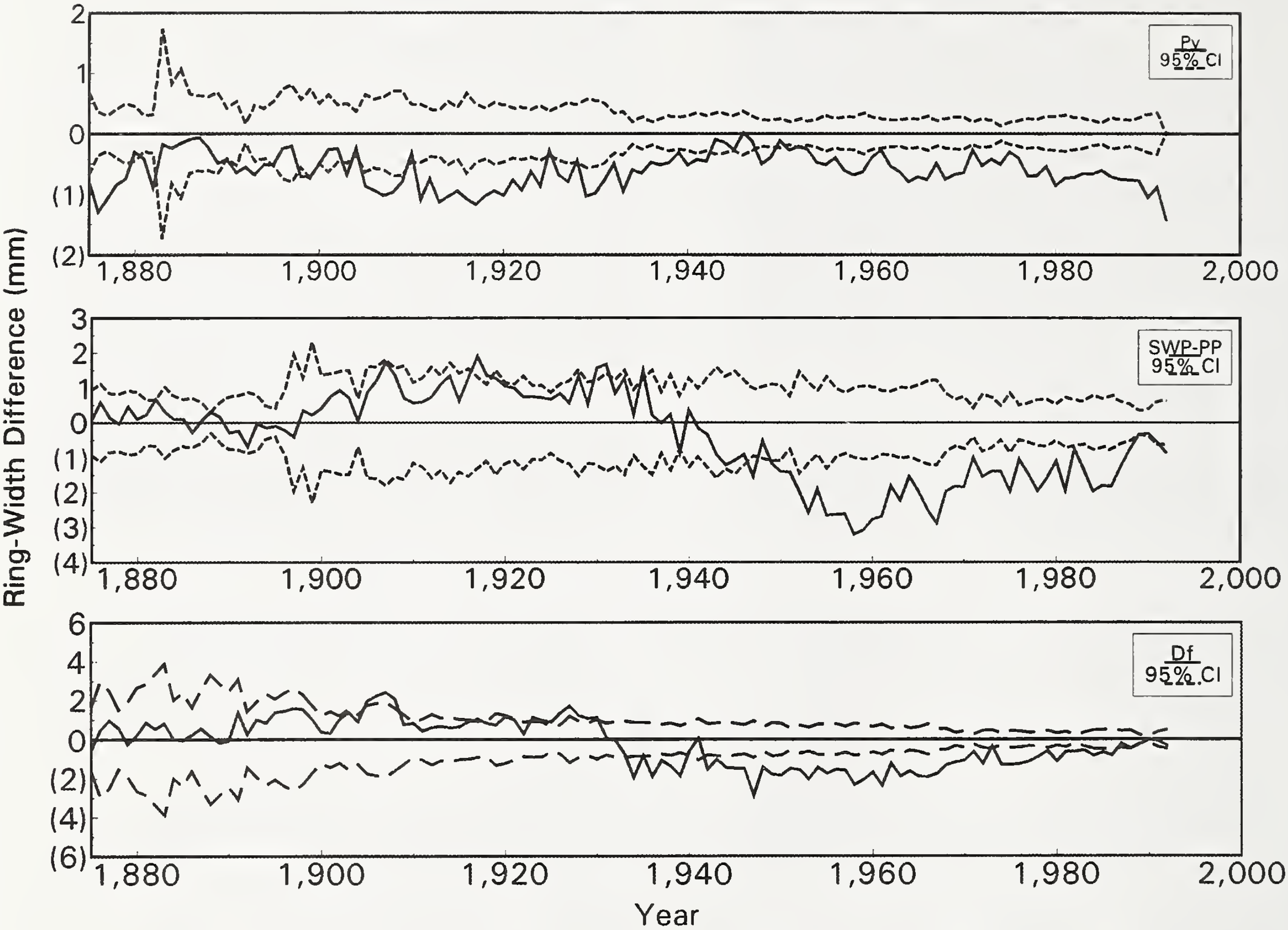

Figure 7. Statistical t-test for comparing average ring-width growth differences for similar taxa between mountain ranges

causing increased intraspecific competition and lower growth rates. In contrast, relatively frequent fires in SLA during this century may have maintained low densities of pinyon pine trees and subsequently higher rates of growth.

Southwestern white pine in the AM and ponderosa pine in the SLA had similar growth rates before about 1900. For the next 40 years average growth rate of southwestern white pine in the AM was greater than for ponderosa pine in the SLA. Southwestern white pine growth slowed in the 1940s relative to ponderosa pine in the SLA, and ponderosa pine growth exceed southwestern white pine growth after 1950. We hypothesize that dominance of young southwestern white pine trees starting in the 1900s in the AM was due to fire suppression. However, increased competition, aging, and low precipitation during the 1940s and 1950s, caused decreased growth. On the other hand, ponderosa pine stands in SLA apparently were harvested for timber on the 1930s and 1940s (Garza-Salazar 1993). Selective timber harvesting and continuation of fires may have maintained lower tree densities, thus favoring an increase in growth. Similar patterns for Douglas-fir growth in the AM and the SLA may reflect these differences in land use: fire suppression in the AM produced substantial recruitment of shade-tolerant Douglas-fir seedlings, and rapid growth of these young trees occurred during 1900-1935, after the 1930s increased competition, higher rates of aging, and low precipitation contributed to decrease growth of Douglas-fir in the AM. In contrast, increased growth of Douglas-fir in the SLA after 1935 may have resulted from timber-harvesting and frequent fires.

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The Role of Fire in Madrean Encinal Oak and Pinyon-Juniper Woodland Development

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Abstract.—Fire has played an important role in the ecology of the oak and pinyon-juniper woodlands of the Madrean Archipelago. However, the nature of fire has changed since intensive European-American settlement at the end of the 19th century, altering the characteristics of the woodlands. This paper reviews the historical role of fire in the encinal and pinyon-juniper woodlands, comparing what is understood about the historical vegetation structure, how fire and/or the lack of fire changed these two ecosystems, and especially how current conditions compare with the historical vegetation structures. It examines the current use of fire in land management and the possibility of further reintroduction of fire into the ecosystems as a way of improving ecosystem management. Knowledge gaps and areas of additional research are identified.

INTRODUCTION

Most ecologists, land managers, and resource specialists agree that fire was a common and powerful disturbance (Fulé and Covington 1995) that historically has been a significant ecological factor in the development of all southeastern Arizona ecosystems (Allen 1994). Whereas the prevalence and occurrence of fire prior to rapid European-American settlement in the last century has been recognized (Caprio and Zwolinski 1995), many also agree that the subsequent exclusion of fire over the past 90 years has resulted in dramatic ecological changes in these ecosystems as well. Leopold (1924), Whittaker and Niering (1964), and, more recently, Bennett and Kunzmann (1992) and Caprio and Zwolinski (1995) suggest that fire plays an important role in determining community composition and diversity, natural selection processes, population dynamics, and species distributions in these ecosystems. There is a growing agreement that the dense stand conditions and changes in relative species composition have developed because of the exclusion of fire, thus increasing the chances of major stand replacing fires or insect and disease infestations. Managers are considering the use of prescribed burning or managed natural fires. However, there seems to be some uncertainty about the specific effects of fire and how to manage it in the Madrean woodland and forests of the Southwest. The role and effects of fire have been studied in the pinyon-juniper woodlands of northern Arizona and New Mexico (Wright and Bailey 1982, Gottfried et al. 1995b); however, there has been relatively little research in the Madrean evergreen woodlands (Brown 1982, Caprio and Zwolinski 1992). One controversy among scientists, managers, and ecologists has to do with the historical role and effects of fire in these ecosystems versus the anthropological effects on fire regimes.

GENERAL DISTRIBUTION AND PHYSICAL ENVIRONMENT

Encinal Oak Woodland

Oak woodlands are common and characteristic of the Madrean Archipelago of southeastern Arizona, southwestern New Mexico, and northern Mexico. They are an important source for natural resource

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products and amenities, including recreation, watershed protection, landscape diversity, and habitat for a number of rare, endangered, and sensitive wildlife species. Encinal and Mexican oak-pine woodlands cover approximately 870,000 ha in the region (Hendricks 1985); about 445,000 ha occur in Arizona (McClaran et al. 1992), primarily in the area from the international border to the Gila River. The oak woodlands are found at intermediate elevations between lower, more xeric, semi-desert grasslands and the higher, more mesic, oak-pine woodlands. Niering and Lowe (1984) report that the encinal region extends from 1,400 to 2,130 m in the Santa Catalina Mountains. Precipitation and tree stand densities increase with elevation (Whittaker and Niering 1964). Precipitation has a distinct bimodal distribution of wet winters and summers. Summer precipitation during the May through August growing season may account for as much as 70 percent of the annual precipitation in some areas (Bahre 1991). Mean annual precipitation within the Madrean evergreen woodland usually exceeds 400 mm (Brown 1982), with a normal range of between 300 and 600 mm (Niering and Lowe 1984). Soils within the Madrean woodland are generally deep, gravelly and cobbly, moderately coarse to fine textured (Hendricks 1985), and soil development depends on parent material and erosional surface stability.

Pinyon-Juniper Woodlands

The identification of distinct pinyon-juniper woodlands in the Madrean Archipelago has been inconsistent. Neither Whittaker and Niering (1964) nor Niering and Lowe (1984) identified a separate pinyon-juniper community. Niering and Lowe (1984) included the pinyon-juniper woodlands in the *Pinus cembroides*/oak woodland and pygmy conifer/oak scrub components of the upper encinal woodland. They identified the woodland community on relatively drier sites between 1,830 and 2,130 m in the Santa Catalina Mountains while the scrub community occurred on steeper, open rocky slopes and crests. However, Pollisco et al. (1995) found numerous sites within three other mountain ranges in southeastern Arizona where a distinct pinyon-juniper community could be identified based on relative basal area. Oaks were common understory species within most stands. The pinyon-juniper woodlands of the Madrean Province generally occur in a band between the oak woodlands and the higher elevation ponderosa pine (*Pinus ponderosa*) or Arizona pine (*P. ponderosa* var. *arizonica*) forests. They are found below the*P. chihuahuana* (=*P. leiophylla var. chihuahuana*)/ oak woodland community in the Santa Catalina Mountains (Niering and Lowe 1984). They grade into the encinal oak associations to the extent that there is a growing concern about pinyon-juniper encroachment into the more commercially important oak woodlands (Pollisco et al. 1995). There is a general lack of information about factors influencing the overlap areas although important physiographic, soil, and edaphic characteristics are similar for both woodland types. In the Madrean woodland provinces, climatic (temperature and precipitation regimes) and elevational gradients are similar between the encinal oak and pinyon-juniper types.

VEGETATION COMPOSITION AND STRUCTURE

Encinal Woodlands

Brown (1982) describes that this mild winter/wet summer woodland is centered in the Sierra Madre of Mexico and reaches northward to the mountains of southeastern Arizona and southwestern New Mexico. He describes the trees as evergreen oaks, deciduous oaks, alligator bark (*Juniperus deppeana*) and one seed (*J. monosperma*) junipers, and Mexican Pinyon (*P. cembroides*) in unequal proportions. Principal encinal oak species are Emory oak (*Quercus emoryi*), Arizona white oak (*Q. arizonica*), and Mexican blue oak (*Q. oblongifolia*). These oak species are relatively small, often multiple-stemmed, and irregularly formed, trees. Species compositions and stand densities depend largely upon specific site characteristics (Gottfried and Ffolliott 1993). One-, two-, or occasionally three-aged stand structures are found. Stand density is related to soil properties, site characteristics, and fire and land use histories (Gottfried et al. 1995a).

Intermingled with these trees are shrubs, grasses, forbs and succulents, often in parks or savanna-like mosaics. Prevalent grass species include side-oats grama (*Bouteloua curtipedula*), the muhlys (*Mulenbergia emerslyei, M. torreyi, M. porteri*); woolspike (*Elyonurus barbiculmis*); and cane bluestem (*Bothriochloa barbinodis*). Some herbaceous forbs and shrubs in-

clude the penstemons (*Penstemon* spp.), lupine (*Lupinus* spp.), bricklebushes (*Brickellia* spp.), sages (*Salvia*), indigobushes (*Dalea*), buckwheats (*Eriogonum* spp.) and many others. The presence of these species, trees and grasses included, varies from occasional to dominant across the biotic spectrum of the Madrean habitat types.

Madrean Pinyon-Juniper Woodlands

The pinyon-juniper woodlands of the Madrean Province differ from those in the northern areas in species composition and are adapted to warmer winter temperatures and greater amounts of precipitation in the July through September period (Hendricks 1985). Border pinyon (*P. discolor*), identified as Mexican pinyon in some literature, is the most common pinyon, although a single-needle pinyon (*P. californiarum* var. *fallax*) is found in parts of the region. Mexican pinyon is widespread at lower elevations in the mountains of northern Mexico (Gottfried et al. 1995a). Alligator juniper is the most common juniper in the Madrean Archipelago and in northern Mexico.

Pinyon-juniper woodlands are not homogeneous and consist of a large number of habitat types or plant associations (Moir and Carleton 1987). The USDA Forest Service (1987), for example, recognizes six habitat types in southern Arizona and New Mexico where border pinyon dominates and six where either alligator or redberry junipers dominate (Gottfried et al 1995a).

Humphrey (1962) suggests that junipers have invaded grassland and the "openess" of the juniper areas depends on soil type, range history, and condition. Due to this association with grasslands, the understory component of these habitat types principally includes blue grama (*Bouteloua gracilis*), pinyon ricegrass (*Piptochaetium fimbriatium*), several muhlys (*Muhlenbergia* spp.), dropseeds (*Sporobolus* spp.), and junegrass (*Koeleria cristata*).

Herbs commonly found throughout much of the woodland include gilias (*Gilia* spp.), buckwheats (*Eriogonum* spp.), penstemons (*Penstemon* spp.), several globemallows (*Sphaeralcea* spp.), and lupines (*Lupinus* spp.).

Indigenous to some habitats and often important subdominant associates, are cliffrose (*Cowania mexicana*), Apache plume (*Fallugia paradoxa*), barberry or algerita (*Berberis* spp.), and fourwing saltbush (*Atriplex canescens*). Several succulents include the hedgehogs (*Echinocereus* spp.) and prickly-pear (*Opuntia* spp.) cacti. Common throughout most of the habitat types within the Madrean Province ecosystem are *Yucca* spp., sotols (*Dasylirion wheeleri*), and several agaves (*Agave* spp.).

THE ROLE OF FIRE

Natural Fire

Fires generally occurred during the dry period between May and late July (Swetnam et al. 1992). Many of the important oak and associated understory species are adapted to surviving fires; Emory oak and Mexican blue oak, for example, sprout after their tops have been killed (Caprio and Zwolinski 1992, 1995). In pre-European times, fire may have burned for months at a time and covered thousands of hectares (Swetnam 1988). Mean fire frequency in the encinal evergreen woodland is unclear and speculative. Niering and Lowe (1984) indicate that the natural open oak woodland, like the adjacent semi-desert grassland, is capable of carrying fire. Whereas fire is generally acknowledged to have been present and influential in the upper encinal or Mexican pine-oak woodland (Leopold 1924, Marshall 1963), the logic developed concerning fire in this community has been based on subjective analysis (Severson and Medina 1983). The lower elevation encinal sites, however, appeared relatively little changed (Bock and Bock 1988).

It is usually assumed that fire played an important role in the Madrean conifer woodlands as well. Niering and Lowe (1984) based their impression on the signs of past fires in the *Pinus cembroides*/oak community. Fire frequencies were probably similar to those of the adjacent oak woodlands.

The effects of fire on trees is influenced by size of tree, amount of herbaceous fuel, wind speed, air temperature, stand density, vertical and horizontal fuel distribution, and season (Moir 1982, Pieper and Wittie 1988). Fire impacted species composition and density. While Bennett and Kunzmann (1988) suggest pinyon is not well adapted to fire and could be damaged or killed depending on the type of fire and heat intensities, Moir (1982) suggests that low intensity surface fires may not damage pinyons and that as these survivors grow to larger sizes, mortality de-

creases. Barton (1993) found higher densities of young trees than mature trees at higher elevations in the Chiricahua Mountains. He indicates that pinyon may not reach maturity on these sites in spite of the ability to become established, because slow juvenile growth and thin bark would make young pinyons very susceptible to damage by periodic fires. Alligator juniper is noted for its ability to sprout after fires or harvesting (Gottfried 1992) and appears adapted to fire, while redberry or one-seed juniper are not considered to be sprouting species. However, young junipers are susceptible to fire. In an Arizona study, 70-100 percent of the one-seed juniper trees less than 4 feet in height were killed by fire (Pieper and Wittie 1988). Moir (1982) suggests that periodic fires seem an important mortality factor in reducing the component of young trees.

Anthropogenically-induced Changes With Fire and Climate

Given that there was some interval of 'natural' fire in these Madrean encinal and pinyon-juniper woodlands prior to settlement, changes in the characteristics and timing of 'natural' fires affect the current distribution of vegetation zones (Gottfried et al. 1995a). Anthropogenically induced fire effects on vegetation result from two sources: 1) fire suppression, either the physical control of fires by federal and state agencies, or, the modification of fuels by grazing and other causes (Allen 1994), and 2) the proliferation of human-caused fire, either by the original Americans or by present day managers (prescribed fire), carelessness, and/or arsonists.

Evidence within the Madrean encinal woodland suggest that vegetation changes have occurred during the last 100 years (Caprio and Zwolinski 1995). Historical over-grazing of livestock has been one factor affecting the role of fire in the region (Gottfried 1992). The reduced and spatially discontinuous herbaceous cover is no longer able to carry fires through the woodlands, allowing woody vegetation to increase in density. The reduction in competition between trees and herbaceous species for moisture, light, and soil nutrients also has favored tree establishment. Livestock also transport juniper and mesquite seeds throughout the area.

The ecological consequences of many years of fire suppression are another factor that has significant impacts on the structure of many ecosystems (Wright and Bailey 1982, Fulé and Covington 1995). Many upper elevation evergreen woodland sites are more heavily wooded today than they were in the 1890's, which Humphrey (1987) attributes, at least partially, to fire suppression policies that were in place over the past 90 years. Several authorities, including Marshall in 1957 (Bennett and Kunzmann 1992), note that similar stands in Mexico, which has less aggressive fire suppression, were more open and less shrubby than those in the United States.

FIRE AND WOODLAND MANAGEMENT

Fuelwood Harvesting and Fire

The primary consumptive use of the encinal woodlands in the southwestern United States and northern Mexico is fuelwood. Fuelwood has been harvested in this region for several centuries and used as a source of fuel for heating and cooking. Since European settlement, the encinal woodlands have been utilized to support the mining industry and ranching, as well. In the 1970s, a sharp increase in human population occurred in the Southwest which was coupled with a severe global fuel oil shortage. These created a renewed increase in demand for fuelwood for home heating in the region.

Because of the large demand placed on woodlands located near small cities and metropolitan areas, large portions of the encinal woodlands in the region have been, and continue to be, harvested. They are primarily areas which contain easy access, mature trees, and minimal slope. Management practices on National Forests in southwestern New Mexico and southeastern Arizona dictate the use of seed tree, single tree selection, group selection, and minor clearcut techniques for harvesting. Harvesting is accomplished by commercial and/or non-commercial woodcutters during specific seasons or on a year-round basis.

The abundant sprouting that occurs after the encinal oaks have been harvested or damaged by fire indicates that coppice methods are appropriate to these woodlands (Gottfried and Ffolliott 1993). Furthermore, these studies indicate that harvesting cycles can be reduced by proper thinning of the resulting coppice. It is recommended that sprouts be thinned to retain one to three of the most vigorous plantlets five years after the harvest (Gottfried and Ffolliott

1993). After these dominant sprouts begin to out-compete the others, the number of sprouts on the stump dwindles. This process may take ten to twenty years to occur. Thinning of coppice after harvesting to reduce competition among sprouts will reduce the time between rotations in the stand (Bennett, 1990). Thinning on some sites can result in trees growing to 15 to 20 cm diameter at root collar in 20 to 30 years. However, more recent studies have indicated that when thinning occurs on sprouts of four and five years of age, additional sprouts emerge on the stump within one year (Bennett, 1990).

Once harvested, the woodlands become thick with slash, and later, sprouts, where trees were cut. It may take more than a year for the leaves to fall from the branches, and more than ten years for the barren branches to disintegrate. Due to the branching characteristics of oaks in this region, the slash left after a harvest makes it extremely difficult to maneuver through. Typically, slash heights in fuelwood areas are about one meter; this is fairly constant throughout the entire area.

Historically, prescribed broadcast fire has been used in these areas to reduce the fuel loading and to clear the brush for cattle grazing. Annually, managers would plan large prescribed fires for fuelwood areas harvested a year or two previously. Managers soon realized that for several years after a harvest the slash was highly flammable and that prescribed fires in these areas were actually damaging the woodlands. Branches with dried leaves were not only extremely flammable, but they created a ladder which enabled the fire to climb into tree crowns. Often, the residual live trees, which were surrounded by brush, were scorched or completely burned.

An alternative to broadcast burning is piling and burning. Although this more manageable option reduces the threat to residual trees, the fire intensity becomes so great that soil beneath the piles is sterilized for many years. This method also proves increasingly costly to implement over large harvested areas.

Management of Wildfire in the Encinal Woodlands

For many decades, policy regarding wildfires on Federal, State, County and private lands in the southwest has been one of total suppression. Even after the realization that these woodlands have developed through time with fire, land managers were told to suppress even the smallest wildfire. Within the last decade, however, some managers have been given the opportunity to monitor fires in certain locations. Generally, these were areas of relatively "low resource value". This policy allows wildfires which do not threaten human life and habitations to burn. The latter are monitored to avoid the possibilities of problems that could result in major suppression efforts. Presently, however, all wildfires that start in higher resource value sites or within urban interface areas of the forest are attacked quickly. However, the re-introduction of fire is controversial.

In Mexico, the Secretary of Environment, Natural Resources, and Fisheries (SEMARNAP) is the federal institution that dictates the rules and procedures for preventing, fighting, and controlling wildfires throughout the Natural Resources Division (Branch). The SEMARNAP functions are to supervise, coordinate, and set actions for fire prevention and for fighting wildland fires, either human-caused or natural caused. In addition, the SEMARNAP promotes technical assistance and organizes educational campaigns on fire prevention and the control of wildland fires. In northeast Sonora, for example, several projects have been proposed and developed by state government institutions, such as the Institute of Ecology and Secretary of Agriculture and Water Resources, in order to manage and protect natural resources (e.g. forest communities, wildlife, water).

Fire Management in Pinyon-juniper Woodlands

Fire could be used in conjunction with silvicultural operations within these woodlands. Silvicultural prescriptions currently are being developed for the southwestern pinyon-juniper woodlands, although, none are specifically designed for conditions within Madrean stands. Selection and shelterwood methods are recommended for the tree regeneration on productive woodland sites (Gottfried and Ffolliott 1993). Overstory removal or simulated shelterwood prescriptions have been applied to stands with sufficient advance regeneration. The seed-tree and clearcut methods, which result in limited seed dispersal, will not result in satisfactory regeneration except where alligator juniper predominates. Fire has been used to dispose of harvest slash or to reduce tree densities. As with oak woodlands, care must be exercised to

avoid damaging residual trees, including advanced regeneration. Silviculture and fire prescriptions must be related to habitat conditions.

Fire has been used on drier southeastern Arizona sites where one-seed juniper and mesquite occur together in relatively open stands. The objective was to reduce tree densities and to encourage herbaceous species. The fires were ignited using both aerial and ground techniques. In some cases the uneven distribution of fuels and of firing success have resulted in mosaics of grass and tree dominated zones. Mosaics are beneficial for a range of animal species and for landscape diversity. Prescribed burning also allows for greater protection of critical riparian areas.

In Mexico, the importance of the role of fire involves projects in both the upper and lower elevation woodland and forest communities. Little information is included with relation to the *natural* role of fire in preserving forest dynamics and nutrient cycling. Projects related to forest preservation such as "The Administrative Proposal for Natural Resources Preservation for the Mariquita, Buenos Aires, and La Purica Mountains" are focussed on fire prevention as a measure for the preservation of plant communities and, again, the importance of natural fires for tree species survival is neglected. Generally, all levels of management in Mexico consider fire as negative for ecological, economical, or social reasons and therefore, to minimize damage to the natural resources all fires should be immediately suppressed.

FUTURE RESEARCH

The importance of expanding fire related research in the Madrean Province of the United States and Mexico has been recognized (Ffolliott et al. 1992) and new projects are being developed and implemented. Present research being done in the mountains of southeastern Sonora, for example, indicates that species composition has been influenced significantly by fire. These studies suggest that fire suppression may not be the best alternative for plant community preservation in these regions and that controlled fire may be used as an alternative option for promoting seedling recruitment of fire adapted species. More fire frequency studies in these Mexican regions may support land managers with the tools they need to decide whether controlled fires are suitable options to promote natural resource preservation.

In the United States, fire is frequently used to reduce stand densities in woodlands or to eradicate trees from grasslands or savannas. Bennett and Kunzmann (1992) caution about this approach in the Madrean evergreen woodland because managers do not have the ecological knowledge to predict the consequences. The 1992 symposium on the ecology and management of oak and associated woodlands in Sierra Vista (Ffolliott et al. 1992) recommended increased fire related research: to describe natural fire regimes in relation to the health of the ecosystem; determine the impacts of fire on threatened, endangered, and sensitive species; determine potential use of prescribed fire to meet land management objectives; and broaden the inventory and monitoring of effects of management actions. There was interest in research to evaluate successional dynamics in stands with and without fire and other disturbances.

Within these broad categories, some specific concerns are: to examine the fire effects on stand composition and structure, to evaluate the possibility of fire creating more fire-tolerant plant communities, or to determine if fire would reduce or eliminate fire-sensitive species from the ecosystem. Also, what are the effects of prescribed burning during the late spring early summer dry season versus the cooler and more moist late summer and fall? Bennett and Kunzmann (1992) also question the impacts on vertebrates, especially the avifauna of the Madrean woodland, and advocate more research on fire effects before a program is initiated. They imply that the absence of fire signs may indicate that fires were not frequent occurrences and that the present stands are the normal natural condition.

Research in ecosystem management should examine the suitability of making broad ecological assessments or inventories of encinal and pinyon-juniper woodland habitat types from a landscape perspective. Landscapes comprised of different mosaics, and specifically fire influenced mosaics, increase diversity for a greater number of wildlife species or classes than landscapes without those characteristics. Quantifying different mosaics, burned as well as unburned, within a woodland type relative to tree species, stand density, understory component could describe:

1. The successional stage of the mosaic;
2. The quantity of wildlife habitats within the mosaic relative to the successional stage; and
3. The quality of the mosaic relative to wildlife habitat diversity (Kruse 1992).

Identifying unique characteristics (i.e. snags, springs, slope, rock outcrops, etc.) would be useful in understanding the dynamics involved in attaining an anticipated desired future condition especially where fire, prescribed or natural, is involved.

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Nutrients in Fire-Dominated Ecosystems

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Abstract.—Fires can produce a wide range of changes in nutrient cycles of forest, shrub, and grassland ecosystems depending on fire severity, fire frequency, vegetation, and climate. These changes can be beneficial when fires increase the availability of plant nutrients, and deleterious when they volatilize, entrain ash in smoke columns, increase runoff of mineralized nutrients, or accelerate leaching from soil systems. This paper examines the effect of fires on nutrient cycling and flux pathways with special emphasis on Madrean-type ecosystems. Most of the best information on fire impacts on these ecosystems comes from research done in interior and coastal chaparral ecosystems north and west of the Madrean floristic province.

INTRODUCTION

Fire is a physical-chemical process that occurs as a continuum of intensities and severities across small areas and whole landscapes. The severity of its impacts depends upon the interactions of fire temperature, duration, combustion phase, fuel load, vegetation, climate, slope, topography, soils, and area burned. Thus the impact of forest, shrub, and grass fires on nutrient resources varies over a continuum. Our ability to describe those impacts is a function of scientific information obtained over a rather limited range of fire intensities, knowledge of nutrient cycling processes, and understanding of fire as a physical-chemical process.

Fire intensity refers to the rate at which a fire produces thermal energy in the fuel-climate environment in which it occurs. It can be measured in terms of temperature, heat yield, and rate of spread. Temperatures can range from 50 to 1,000° C, and heat yields can be as little as 2 BTU/kg or as high as 2,000 BTU/kg. Rates of spread can vary from 0.5 m /week in peat fires to as much as 7 km/hr in large wildfires. The intensity produced by any fire is an integration of the fuel conditions and climatic conditions that precede ignition. Fuel loadings range from <1 Mg/ha in grasslands to 400 Mg/ha in denser old-growth tree stands. Fuel moistures and organic constituents play a prominent role in the rate at which fires release thermal energy. Finally, fire intensity is determined by the interactions of climate conditions such as air temperature, relative humidity, antecedent rainfall, and wind. Slope, topography, altitude, soils, and fire size may further modify intensity.

The facet of fire intensity that results in the most damage (fire severity) to site nutrient resources is duration. Because fast moving fires in fine fuels, such as grass, do not transfer as much heat to a forest floor or mineral soil as do slow moving fires in heavy fuels, their impacts on nutrient cycling processes are much less severe. Severity is a qualitative measure of the effects of fire on soil and site resources (Hartford and Fransen 1992).

Heavy fuels with loads of >100 Mg/ha can produce temperatures in excess of 1,000° C. On the other end of the fuel spectrum, grass fires usually have temperatures of 80 to 160° C. Fire-prone vegetation types, such as chaparral, burn at temperatures of 350 to 370° C (Woodmansee and Wallach 1981). At temperatures above 300° C, the entire forest floor is usually consumed. In the range of 180 to 300° C, 50% of the litter layer is burned. Litter is only scorched at temperatures under 180° C.

Soil temperature effects on nutrient cycling processes are particularly important since they quickly

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affect the ability of forest ecosystems to recover and sequester nutrients released by the combustion of above ground organic matter (Hungerford 1996). Disruption of soil nutrient cycling processes is then a function of the severity of burning. Temperature profiles in the mineral soil depend on the intensity of the fire, fuel loads, duration of the burning, and antecedent soil moisture (Hartford and Franson 1992, Hungerford 1996). With less severe soil heating, mineral soil temperatures usually do not exceed 50 Co at 1 cm. However with severe soil heating, temperatures can reach temperatures >250° C at depths of 9-16 cm, causing considerable mortality to soil organisms and volatilizing nutrients. Disruptions begin in the 40-70° C range with plant tissue death. At soil temperatures of 50-56° C, roots are killed; seed mortality occurs in the 70-120° C range. Although most microorganisms die off between 50-160° C, fungi are less resistant to thermal effects than bacteria and hyphal mortality happens at the lower end of this temperature range. Organic matter distillation starts in the 200-315° C range, and volatilization begins when temperatures climb to 350-400° C.

An important component of understanding the effects of fire on nutrient resources is knowledge of nutrient cycling. This includes not only nutrient pools, but also key fluxes, transformations, and losses. Figure 1 is a generalized cycling diagram for nitrogen (N) in a forest ecosystem. The arrows indicate fluxes between pools (square boxes) and losses from the ecosystem (circles or ovals). The arrows indicating

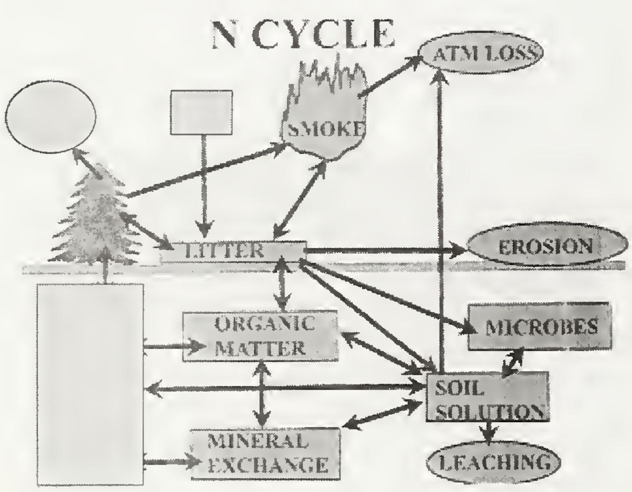

Figure 1. Major pools and flux pathways for nitrogen in fire-dominated forest ecosystems.

fluxes do not indicate their magnitude. Other major nutrients in the organic pool, such as phosphorus (P), sodium (Na), potassium (K), calcium (Ca), and magnesium (Mg) follow the same pathways except that direct loss to the atmosphere from the soil does not occur.. The discussion that follows utilizes Figure 1 as a framework.

Information on the effects of fire on nutrients in Madrean ecosystems is limited and incomplete. Much of the best information on nutrients comes from research in interior chaparral ecosystems to the north of the Madrean floristic province, elsewhere in North America, Australia, and the Mediterranean region. So, some fire effects information from these other regions is incorporated into this analysis to provide a more complete picture of nutrient processes and dynamics in fire-dominated ecosystems like the Madrean floristic province.

NUTRIENT POOLS AND FLUXES

Atmosphere

Global Implications

Burning of fuels in the Madrean floristic province certainly has localized impacts on nutrient cycling. However, it is also part of the annual global biomass burning, which consumes 8,680 million metric tons and produces atmosphere-wide and ecosystem-wide ecological effects (Levine 1991). Of this amount, relative distribution is 42% grass, 18% forest, 23% agricultural waste, and 17% fuelwood. The majority of biomass burning is human caused, and appears to be on the increase. Biomass burning is a major contributor to greenhouse gases, producing 40, 38, 32, 39, and 20% of the world's annual production of carbon dioxide, ozone, carbon monoxide, particulate organic carbon, and other greenhouse gases (hydrogen, nonmethane hydrocarbons, methyl chloride, and nitrogen oxides), respectively. Burning in the Mexican portion of the Madrean province significantly exceeds that of the United States portion (Fule and Covington 1995).

Ash Convection and Wind

Nutrient-rich ash materials remaining after a fire can be removed from or redistributed within a burned

area by convection in a smoke column or surface wind transport. These processes account for a small portion of atmosphere-related nutrient losses during and immediately after small, low-intensity fires, but are a more important consideration after large wildfires and in windy regions. For low-intensity fires, ash lost in convective smoke plumes can amount to 1-4% of the mass of the burned fuel (Raison et al. 1985a). With higher intensity fires, ash losses in convective columns can exceed 11 %. Data on nutrient losses in wind-borne ash after fires are very scarce, but losses are estimated to be low because of the relatively low mass transport of soil and ash in wind from most burned areas (Baker and Jemison 1992).

Volatilization

Nutrients are volatilized directly into the atmosphere if temperatures are high enough. Nitrogen is the element most prone to this type of loss since it starts to vaporize at 300° C. At temperatures above 500° C, over half the N in organic matter can be volatilized. Hotter temperatures are needed to vaporize K (760°+ C), P (774°+ C), sulfur (S) (800°+ C), Na (880°+ C), Mg (1107°+ C) and Ca (1240°+ C) (Weast 1988).

Harwood and Jackson (1975) reported nutrient losses to the atmosphere from low intensity slash burning of 10-17% of total nutrients (P, K, Ca, and Mg) in the fuel. Raison et al. (1985b) measured higher, non-particulate elemental transfers to the atmosphere from low intensity slash burning of 54-75% (N), 37-50% (P), 43-66% (K), 31-34% (Ca), 25-49% (Mg), 25-43% (manganese -Mn), and 35-54% (boron - B). Grier (1975) documented nutrient losses to the atmosphere through volatilization and ash convection from the Entiat wildfire in Washington (mixed conifer forest). Most of the N loss (39%) was volatilization, as expected, but much of the monovalent cation losses (35% for K and 83% for Na) could have also been through volatilization due to the high fire intensity.

Litter

The litter layer is an important component of forest ecosystems in that, among other things, it provides a cover to forest soils to mitigate the erosive effects of rainfall, and is a source of mineralizable nutrients. It consists of the organic rich horizons L, F, and H horizons, collectively designated the "O" (organic) horizon. The L layer consists of freshly fallen plant material. Beneath this is the F (or fermentation) layer of partially decomposed, but mainly recognizable plant material. The final litter layer above the first mineral soil horizon is the H (humus) layer of highly decomposed plant and animal-derived organic matter consisting of carbohydrates, amino acids, proteins, nucleic acids, lipids, lignins, humus, and minerals, listed in order of resistance to decay (McColl and Gressel 1995). Prior to incorporation in the mineral soil through animal activity, organic matter is particularly susceptible to combustion in fires and loss of key nutrients. Depending on variations of fire severity, litter consumption can range from scorching to complete ashing.

The amount of litter in an ecosystem and its importance in nutrient cycling are dependent on the plant type (grass-shrub-forest), productivity, climate, and decomposition rates. Tiedemann (1987) contrasted biomass and nitrogen on a shrub-grass-forb to sagebrush-grass to pinyon-juniper ecotone in the intermountain region of the West (Table 1). Across this

Table 1. Biomass and N distribution: shrub-grass, sagebrush, and pinyon-juniper ecosystems (from Tiedemann 1987).

Component	Biomass (mg/ha)	Nitrogen (mg/ha)	N %
LOW SHRUB			
Overstory	——	——	——
Understory	0.600	0.006	0.1
Litter	0.300	0.002	0.1
Soil (0-22 cm)	——	4.200	99.6
Roots	1.200	0.012	0.2
TOTAL	2.100	4.220	
SAGEBRUSH			
Overstory	11.400	0.059	1.3
Understory	1.100	0.032	0.7
Litter	5.450	0.039	0.8
Soil (0-60 cm)	——	4.527	94.1
Roots	16.800	0.150	3.0
TOTAL	34.750	4.768	
PINYON-JUNIPER			
Overstory	97.000	0.425	5.2
Understory	0.400	0.004	0.1
Litter	100.000	1.000	12.4
Soil (0-30 cm)	——	6.615	81.9
Roots	36.000	0.030	0.4
TOTAL	233.400	8.074	

gradient, the amount of litter increased from 0.3 to 100.0 Mg/ha. Pase (1972) documented slope-dependent litter layers of 9.2 to 16.4 Mg/ha on north and south slopes, respectively, in unburned chaparral of central Arizona. Wienhold and Klemmedson (1992) found litter masses of 33 Mg/ha under unburned chaparral stands in the Bradshaw Mountains north of Phoenix, Arizona. Litterfall rates of 0.192 to 0.176 Mg/ha/yr were measured. Covington and Sackett (1992) reported litter layer buildups of 27.1, 35.9, and 123.4 Mg/ha in unburned ponderosa pine stands on the Mogollon Rim of Arizona that had not burned in over 106 years.

Thus, as ecosystems transition from grass-dominated to woody plant systems, the litter layer increases substantially and the percent of ecosystem elements, particularly N, tied up in litter and susceptible to loss by fire increases. In the ecosystems referenced by Tiedemann (1987) that amount ranged from 0.1 to 12.4%. DeBano and Conrad (1978) reported a similar figure for ecosystem N in litter (10%) in California chaparral (Table 2). In Arizona chaparral, Wienhold and Klemmedson (1992) found that the N component of litter amounted to 16.7% of the ecosystem N prior to a prescribed fire. Likewise, DeBano and Klopatek (1987) reported that 12% of the N in an Arizona pinyon - juniper ecosystem was tied up in litter (Table 3).

Although the main pool for P, a critical plant nutrient, is in the soil not the litter (94 to 98%, Tables 2 and 3), organic forms of P in the litter are more available to plants. So, severe burning of vegetation and litter doesn't necessarily have the same potential impact on P pools as on N. Thus the impact of complete litter combustion on P cycling can be more severe than what is indicated by the size of the individual nutrient pools. Also, P in soils with high Ca levels can be complexed into non-available forms with detrimental consequences for ecosystem productivity. Since the cycling of P is primarily through the organic-P pools, removal of vegetation all at one time by burning results in depletion of the aboveground P pool at a rate greater than mineral weathering can replace it.

Table 2. Nitrogen and phosphorus distribution in California chaparral and loss due to prescribed fire (from DeBano and Conrad 1978).

	Nitrogen			Phosphorus		
Component	mg/ha	%	Fire loss mg/ha	mg/ha	%	Fire loss mg/ha
Plants	0.134	9	0.101	0.010	2	0.009
Litter	0.147	10	0.008	0.022	4	+0.009
Soil: 0-2 cm	0.337	24	0.036	0.120	21	0.102
Soil: 2-10 cm	0.806	57	*1	0.416	73	*
TOTAL	1.424		0.145	0.568		0.030

[1] *Not determined*

Table 3. Nitrogen and phosphorus distribution in Arizona pinyon-juniper (from DeBano and Klopatek 1987).

	Nitrogen		Phosphorus	
Component	mg/ha	Percent	mg/ha	Percent
Plants	0.425	5	0.036	1
Litter	1.000	12	0.044	1
Soil	6.615	82	3.963	98
TOTAL	8.040		4.043	

Litter layer consumption can be highly variable within and between fires. Amounts consumed by fires have been reported to be 21 to 50% (prescribed fire, Pase and Lindenmuth 1971), 34 to 80 % (prescribed fire, Covington and Sackett 1992), and 45% (prescribed, DeBano and Conrad 1978). Campbell et al. (1977) described the effects of moderate and severe burning within a ponderosa pine wildfire on the Mogollon Rim of Arizona. Whereas the moderately burned areas still contained 38% vegetation and litter, the severely burned areas had less than 23% vegetation and litter.

Nutrient losses from the litter layer during fires is a function of fire severity. Grier (1975) noted nutrient losses from the litter layer after the intense Entiat wildfire that amounted to 19 (Ca), 21 (K), 31 (Mg), and 97% (both N and Ca) of pre-burn conditions. DeBano and Conrad (1978) measured N and P changes in the litter layer after a prescribed fire in California chaparral. While the N loss totalled 5% of the pre-burn litter N, the P pool in litter increased 43% due to the accretion of ash from the burned vegetation. For the system as a whole, the N loss was 10% (half coming from the litter), and the P loss was 2% (mainly from reductions in soil P in the top 2 cm)

Soils

The process of soil formation involves the transformation of parent material (rock, colluvium, or

alluvium) over time and under the influence of climate, topography, biological processes, and disturbances. Soils develop into distinct profiles and horizons (such as O, A, E, B, C) depending on the combination of soil-forming factors. Many of the key processes involved in nutrient cycling occur in the soil (Figure 1), and the soil is the most important nutrient reservoir in most ecosystems (Table 1).

Physical Processes

Several soil physical processes and properties are affected by fire. These effects can impact both the nutrient status and cycling in Madrean ecosystems. Consumption of litter and soil organic matter by fire can seriously affect infiltration, the moisture holding capacity of soils, and the ability of soil systems to process rainfall. Removal of vegetation and litter by burning can also alter the post-fire thermal regime in the soil that controls microbial processes. Changes in microbial processing of organic matter ultimately affect the plant-available nutrient pool.

When fires burn off vegetation and litter layers that mitigate the impact of rainfall on the soil, the result often is increased runoff, soil dewatering, and ultimately watershed drying. Bare soil surfaces can seal off under the impact of raindrops, resulting in greatly higher rates of surface runoff. The net effect is a reduction in soil moisture and erosion of nutrient-rich ash and A horizon sediments. Apart from the physical removal of nutrients in runoff (discussed in a following section), the drying of soils diminishes the ability of recovering and colonizing microorganisms and plants to cycle nutrients. Adequate soil moisture is necessary to support microbial processes and to maintain a sufficient flow of nutrients to the roots of growing plants (Neary et al. 1990). Desiccation of the soil can then in turn create a negative feedback on nutrient cycling by decreasing plant cover and increasing surface erosion of nutrient-rich sediments.

Another physical response to fires in soils that is linked to the process just discussed above is water repellency (DeBano 1981). This condition develops in the presence of high intensity fires and certain types of organic matter. Water repellency in flat terrain just contributes to soil desiccation, but in steep terrain it can significantly accelerate erosion. It produces both long term effects (mass wasting, soil erosion, site degradation, etc.) and short term effects (increased runoff, ash and soil transport, etc.) that affect nutrient cycling processes. On a large-area basis, nutrients lost in mass wasting and surface erosion related to water repellency (<0.001 Mg/ha) may only be 10% of the total plant-litter-soil losses (DeBano and Conrad 1978). However, plant re-establishment is difficult on areas stripped of their nutrient-upper soil layers.

Thermal regimes of soils can be significantly altered by fires. Soils under closed forest canopies can experience large increases in maximum daily temperatures (20° C) and decreases in minimum night temperatures after fires. These changes can affect the physiology of microorganisms involved in nutrient cycling processes (Klopatek 1995). Thermal properties of soils (as well as other properties) can be altered when high fire temperatures (700-800° C) destroy mineral particles.

Chemical Processes

Heating during fires can significantly alter the nutrient status and nutrient cycling of forest, shrub, and grass ecosystems (DeBano et al. 1979, Dunn and DeBano 1977). Forest soils are more susceptible to alterations in their chemical state since more of the nutrient capital is aboveground and larger biomass accumulations can produce hotter temperatures during wildfires. However, the bulk of their nutrients still reside below the soil surface (Tables 2 and 3). Some of the effects of fire on soil chemistry and nutrient cycling processes include reduction in cation exchange capacity due to organic matter combustion or displacement, changes (both positive and negative) in nutrient availability as a result of increases in soil pH, direct nutrient volatilization, loss of nutrients due to ash and soil particle erosion, changes in nutrient forms to less or more available forms, and increases in leaching due to the disruption of nutrient uptake (Hungerford 1996).

The effects of fire on elemental cycling, particularly N, can be either minor or profound. Apart from the ability of fire to convert N immobilized in surface organic matter (plants and litter) to N gases, most of the N cycling processes in the soil are directly affected by fire. Some of the main processes affecting nitrate N (NO_3-N), for example, are:

a. Immobilization by decomposers

b. Uptake by growing vegetation

c. Denitrification to N gases (NO_2, NO, N_2)

d. Reduction to ammonia N (NH_4-N)

e. Adsorption on organic matter and clay exchange sites

f. Retention in pore spaces

g. Leaching to groundwater

All of these, except Item f., are directly or indirectly affected, mainly through alterations to plant, soil arthropod, and microorganism populations. Oxidation of litter and surface soil organic matter has important effects on N cycling.

Organic matter losses in the litter layer and A horizon can significantly affect the nutrient status of soils, since organic matter is a long-term source of mineralizable nutrients, and is the major factor controlling total cation exchange capacity (organic and clay mineral). Losses of N in the A horizon associated with organic matter combustion can exceed those of the entire litter layer (Table 2, DeBano and Conrad 1978). Organic matter contents of 4 to 6% in an A horizon, can be reduced to less than 1% organic matter.

A soil chemistry change which typically follows fires, but is most pronounced in fires of high intensity, is increased pH. Complete litter layer consumption increases soil alkalinity by releasing substantial amounts of cations. Most well-developed forest soils have pH's of 5.5 to 6.5, but may be higher where parent material (limestone, dolomite, etc.) is a strong factor. Elements such as Ca, Mg, K, S, and N generally become more available at a pH of 7.0, while copper (Cu), cobalt (Co), Mn, and zinc (Zn) are more available at pH's of <6.0. Nitrogen often becomes more available with light fires when it is shifted from organic forms and NO_3-N to NH_4-N (Hungerford 1996).

Specific changes in soil chemistry have been documented by investigations in ponderosa pine, chaparral, and pinyon-juniper in Arizona at the northern edge of the Madrean floristic province. Overby and Perry (1996) measured significant increases in soil NH_4 in the 0 to 10 cm depth, changing from 5.2 mg/kg before a prescribed fire in chaparral to 48.7 mg/kg immediately after the fire. A smaller but significant response was also measured for P (5.4 to 14.6 mg/kg). The soil pH declined slightly but not significantly, and soil organic carbon increased from 4.7 to 5.6%. Similar, but low magnitude, responses were documented in the 10 to 20 cm soil depth. Covington and Sackett (1992) measured substantial increases in inorganic N (mainly NH_4-N) in the upper 15 cm of soils after prescribed fire in ponderosa pine stands. Old growth stands exhibited the largest response, going from 0.002 to 0.036 Mg/ha due to high litter loads.

Recovery of soil nutrient levels can be fairly slow in some ecosystems, particularly those in the southwestern United States and northern Mexico where decomposition rates are slow. Klopatek (1987) determined that N and P concentrations beneath pinyon-juniper canopies, 35 years after a wildfire, had not recovered to levels found in stands that had not burned in 300 years (Table 4). In addition, when contrasting soils beneath burned and unburned stands, two-fold differences were noted in the percent of total N that was mineralized. This difference indicates that mineralization and nitrification may be enhanced by fire disturbance.

Table 4. Nitrogen and phosphorus time trends and mineralization in northern Arizona Pinyon-Juniper 300 and 35 years after burning (from Klopatek 1987)

Site - Years since burn	Sample	PO_4-P mg/kg	NO_3-N mg/kg	NH_4-N mg/kg	Mineralized %
300	Canopy	38.0	4.4	20.1	2.2
	Interspace	6.2	1.1	9.4	4.8
35	Canopy	23.4	0.5	12.1	4.6
	Interspace	10.1	0.7	8.9	10.6

Soil Biota

These organisms play very important roles in the processing, storage, and transformation of nutrients. Their occurrence and population sizes are influenced by a variety of environmental factors such as aeration, pH, moisture, temperature, energy sources, vegetation type and distribution, and by disturbances, such as fire, that affect these factors (Wells et al. 1979). Klopatek and Klopatek (1987) determined that interspace areas within an ungrazed northern Arizona pinyon-juniper stand contained 213,000 nitrifying bacteria/g of soil, while under the tree canopy nitrifying bacteria were found in numbers of only 35,000/g of soil. Conversely, they found more vesicular-arbuscular endomycorrhizae under the canopy than in interspace areas. Whitford (1987)

studied microarthropods, another component of soil fauna, in soils of pinyon-juniper stands in the Chiricahua Mountains of southern Arizona. He found that pinyon pine litter contained 21,550 microarthropods/m^2, juniper litter contained 11,400 microarthropods/m^2, and soils from interspace areas contained 4,064 microarthropods/m^2.

Microorganisms

There is a scarcity of information available about soil microbial populations within ecosystems of the Madrean Archipelago, and how they are influenced by fire. However, fire effects on soil biota can be inferred from research elsewhere. Under the influence of fire, soil microorganism populations and species composition can increase or decrease depending on the intensity of the fire, maximum temperatures, soil water content, duration of heating, and depth of heating (Hungerford 1996). Site conditions (fuel loadings, aspect, elevation, etc.) and weather conditions pre- and post-fire are important since they can influence the nature of the fire and site recovery. Microorganism responses within and between fires on any given site and between sites within the Madrean ecosystem will fall along a continuum determined by factors mentioned above (DeBano et al. 1995). Low intensity, rapidly moving fires, such as in low fuel load grass understories or grasslands, do not have a major effect on microbial populations. High intensity fires with long durations, such as in mixed conifers with high fuel loads, cause the greatest impact on soil microbes.

Fire effects on microorganisms are greatest in the surface 1-2 cm where microorganism populations are most abundant and heating effects are the greatest. The general trend is for microorganism populations to decline significantly after fires, but then expand dramatically above pre-fire levels as nitrifier populations explode with altered nutrient, thermal, and moisture regimes, and regrowth of vegetation.

Soil heating can kill microorganisms directly (50-210° C) or alter their reproductive capabilities (Covington and DeBano 1990). Where temperatures as high as 210° C are needed to kill bacteria in dry soils, moisture in the soil can reduce lethal levels to around 110° C (Wells et al. 1979). Microbial population changes can also be related to changes in the soil environment; for example, heterotrophic bacteria would be impacted by loss of the organic matter food base after a fire (Barbour et al. 1980). Fire can indirectly affect microbial populations by totally or partially oxidizing organic carbon in and above the soil surface. Organic carbon is needed as a substrate and nutrient source by heterotrophic organisms. Soil pH can also increase after a fire in some forest types, affecting many microorganisms that are sensitive to pH changes. However, fire-related pH changes most likely would be minimal in the near-neutral pH soils commonly found in most of the Madrean region (Covington and DeBano 1990).

The reduction or loss of microorganisms will affect nutrient cycling. For example, both *Nitrosomonas spp.* and *Nitrobacter spp.* groups, which are involved with nitrification, are very sensitive to heat (Dunn and DeBano 1977). Increases in nitrification observed in chaparral soils following fires may be related to heterotrophic nitrification by fungi (Wells et al. 1979), and to increased available NH_4-N resulting from mineralized vegetation and litter N (Overby and Perry 1996).

In general, the effect of fire on soil microorganisms in the short-term is to reduce fungi, but increase populations of bacteria and actinomycetes (Barbour et al. 1980). Actinomycetes are, however, less sensitive to heat and desiccation than are bacteria. Species characteristics are important since some organisms can develop spores or other resting stages that are more resistant than the vegetative stages.

Mycorrhizae are soil fungi that form mutualistic relationships with plant roots. They obtain nutrients from host plants and contribute to plant nutrition, particularly P, and water absorption. They also provide some degree of resistance to plant pathogens and enable their host plants to become established and grow in severe habitats (recently burned areas) where they might not normally survive (Atlas and Bartha 1981). There are two main types of mycorrhizae, the endomycorrhizae that penetrate plant roots, and the ectomycorrhizae that form sheaths around plant roots. The latter are usually associated with conifers, oaks, and other angiosperms. Vesicular-arbuscular mycorrhizae (VAM) are a type of endomycorrhizae that are most commonly found in association with plant roots (Atlas and Bartha 1981).

Klopatek et al. (1988) indicated that colonization of VAM was moderately affected at soil temperatures <50° C, significantly reduced between 50 and 60° C, and severely reduced when burning tempera-

tures reached 80 to 90° C. At 94° C there was a total loss of VAM . Although moisture generally aggravates heat damage in soils during fires, they found that damage was greater when the soils were dry than when they were wet. The duration of a fire in the surface organic debris, as well as the temperatures achieved, affects survival of mycorrhizael. In a pinyon-juniper stand, recovery 5 years after a prescribed fire was greater in interspace areas where initial fuel loadings were relatively light than under canopy areas where fuels were heavier (Klopatek 1995). Several studies in the grasslands, where fires may generate less heat and have a short duration, have not demonstrated long-term declines in mycorrhizae (Klopatek 1995). The time period for below ground ecosystems to recover after fires and the long-term consequences on nutrient availability and site productivity after repeated fires are not well known.

Vegetation

Above Ground Biomass

Nutrients in aboveground biomass pools are most obviously and easily affected by fire. Most of the nutrients lost from burned sites originate in vegetation as well as litter, although in some cases the surface layers of soils can contribute significantly to off-site losses. As indicated in Table 1, forests and woodlands have higher biomass loads than shrub-grass ecosystems so they are more sensitive to nutrient losses in fires. Disruptions to root systems can also affect nutrients on burned sites, since slow recovery of vegetation due to destroyed root systems delays root uptake of fire-mobilized nutrients and increases leaching losses.

Roots

A major role in nutrient cycling of forest, shrub, and grassland ecosystems is played by plant roots. Not only do roots take up and store nutrients while plants are active, but they also are sources of organic matter and nutrients and function as important substrates for microorganisms. Davis and Pase (1977) measured root masses of 172.6 kg/m3 in the upper 1.8 m of soil in chaparral - live oak communities in Arizona, at the northern end of the Madrean province. About 75% of this root biomass was in the upper 0.6 m of the soil. Roots of oaks can penetrate to depths of 9 m in some soils and have been measured as deep as 21 m. Although roots are protected somewhat from the heat of fires, intensive fires can kill entire root systems and smolder in roots underground for a year or more. Cessation of uptake due to plant and root death after fires can have a major impact on the cycling of nutrients, particularly N (Vitousek and Melillo 1979). In Arizona chaparral, significant NO_3-N increases have been measured after fires due to release of this form of N from extensive, decaying root systems that were killed by the fires (Davis 1989).

OFF-SITE TRANSPORT

Erosion

The amount of nutrients lost off-site after fires by surface runoff or debris flows is highly variable. Factors such as fire intensity, slope, watershed condition, storm timing, rainfall amount, precipitation intensity, topography, elevation, and slope affect nutrient loss from this process. The Madrean floristic province is probably the most susceptible ecosystem in North America to this type of post-fire nutrient loss because of the geomorphic sensitivity of the landscapes (high mountains, intense rainfalls, shallow soils, intense and frequent fires, slow vegetation recovery rates etc.). Therefore, sediment and nutrient losses after prescribed fires or wildfires are highly variable. Sediment losses from fires in Arizona chaparral have been reported to range from 3.8 to 204.0 Mg/ha (Neary 1996).

Data on erosion loss of nutrients after fires in the Madrean ecosystem are fairly scarce and difficult to

Table 5. Sediment loss, runoff, and organic matter loss from a prescribed fire in California chaparral (from DeBano and Conrad 1976).

Condition	Slope (%)	Debris (mg/ha)	Runoff (m^3/ha)	Organic matter (mg/ha)
Burned	50	7.340	786	0.410
	20	2.848	584	0.312
Control	50	0.210	25	0.007
	20	0.000	3	0.000

Table 6. Nutrient loss in sediment form a California chaparral prescribed fire (from DeBano and Conrad 1976).

Condition	Slope (%)	N (kg/ha)	P (kg/ha)	K (kg/ha)	Ca (kg/ha)
Burned	50	15.1	3.4	19.3	47.4
	20	7.5	1.0	7.6	18.5
Control	50	0.3	0.1	0.5	0.5
	20	0.0	0.0	0.0	0.0

estimate given the highly variable and episodic responses of watersheds to climatic events. DeBano and Conrad (1976) provided some fairly complete information from California chaparral that can be used to illustrate potential Madrean ecosystem responses. They measured significant slope-related watershed responses to prescribed fire in terms of debris production, runoff, and organic matter lost (Table 5). Nutrient losses in mineral sediments and organic debris were 34 to 94 times background levels on the steeper slopes which responded more to the combination of fire and rainfall event (Table 6).

Leaching

Losses of nutrients from burning in the Madrean system are set up by both the mineralization of large amounts of nutrients at one time as well as temporary termination of uptake by plants (Vitousek and Melillo 1979). The magnitude of nutrient losses after fires in the Madrean floristic province is difficult to gage because of the highly variable nature of both fires and climatic events. However, where data do exist, they indicate that Madrean and other southwestern United States ecosystems have the potential for larger leaching responses than elsewhere in terms of concentrations, but not necessarily total mass (Neary 1996).

SUMMARY AND CONCLUSIONS

Prescribed fires and wildfires can produce a considerable range of changes in nutrient cycles of forest, shrub, and grassland ecosystems in the Madrean floristic province, depending on fire intensity, fire frequency, vegetation, and climate. Grassland ecosystems are less susceptible to disruption in nutrient cycling due to generally lower intensity of fires and smaller amounts of nutrient capital in aboveground pools. These changes can be beneficial where prescribed fires increase the availability of plant nutrients and deleterious when severe wildfires produce nutrient volatilization, ash entrainment in smoke columns, increased runoff of mineralized nutrients, and accelerated leaching from soil systems. More information on the effects of fire on ecosystem nutrients is most available in the smaller United States portion of the Madrean province, where fire frequency currently is lower than the Mexico segment.

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The Effects of Fire on Interpretations of the Past

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Abstract.—The effects of fire on material evidence of past human cultures have not been systematically investigated in the Madrean Archipelago. The potential of fire to alter interpretations of prehistoric and historic human occupation is an important consideration in the sky island region where allowing fire to play a more natural role is a major emphasis on public and private lands. Data from fire effects studies elsewhere in the American Southwest are described and recommendations made for research needs relevant to fire effects on cultural resources.

INTRODUCTION

Fire, as a tool for people to control and modify their surroundings and to assist in various aspects of everyday life, has been used for thousands of years (Deneven 1992). Prehistoric people used fire in a variety of ways which have been documented archaeologically and through analogies with historic Indian societies (Williams 1994). In addition to deliberate use of fire, fire played a more natural role in the evolution of ecosystems because previous societies did not possess the technology, the livestock, nor the fire suppression mandate that has existed during the last century. Therefore, we know that prehistoric settlements burned, that landscapes surrounding these settlements burned, and that much of the mountain and valley environment of southern Arizona and northern Mexico has been exposed to fire at one time or another (Bahre 1991, Caprio and Zwolinski 1995, Grissino-Mayer, Baisan and Swetnam 1995).

Fire, as it relates to cultural resources management can be examined from two perspectives. First is the effect that past societies may have had on their environment through the use fire. Second is the effect that post-occupational fires (prehistoric, historic and modern) may have on the material evidence of prior occupations. In this paper, I focus on the latter. Because there have been no systematic cultural resources and fire effects studies carried out in the Madrean Archipelago I can only make comparisons with similar situations in other parts of the Southwest which may serve as a basis for future studies in the sky island region.

Evaluating the potential effects of fire on material remains of the past is perhaps one of the most challenging aspects of fire planning and cultural resources management. Because we have very limited information, archaeologists may be significantly underestimating or overestimating the impacts of fire on cultural resources. We tend to assume that the cooler temperatures of prescribed burns have much less impact than wildfires, and in wildfire situations the focus has been on reducing impacts from the suppression related activities of use of bulldozers, construction of fire lines, and post-fire mop up work rather than on the fire itself. Systematic and objective techniques for describing impacts to cultural resources have not been consistently used nor have they been adequately tested.

When I speak of "impacts" this is not necessarily synonymous with "damage" to cultural resources. In most cases, we do not know at what point impacts to artifacts, sites and other material remains, damage these resources to the critical point that it affects our interpretations of the archaeological record. The focus here is on the potential impacts of fire itself rather than on fire suppression and management activities that may also impact cultural resources.

I discuss four aspects of fire and cultural resources management:

1. What are cultural resources and what characteristics do they possess relevant to fire?

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2. How do federal agencies such as the Forest Service currently deal with planned fire situations?
3. What data are available regarding the effects of fire on cultural resources?
4. What recommendations can be made for research involving fire and cultural resources in the sky islands? That is, what are the main resource protection issues involved, and what factors need to be considered as we develop fire prescriptions and plan to burn larger expanses of forest and shrubland.

CHARACTERISTICS OF CULTURAL RESOURCES

Cultural resources are the material evidence of past human use. These material remains are valuable because they provide information essential to our understanding of past societies. Archaeological goals are to reconstruct prehistoric human behavior and lifeways, and to understand the processes of cultural change through time. Archaeologists examine and analyze artifacts, features and sites. We also rely on specialists in other fields in developing reconstructions of the past. As examples, soil formation and sediment information are essential to reconstruct geological processes relevant to human environments; faunal remains and floral elements from pollen and plant macrofossils are examined to provide subsistence data, and climatic reconstructions may rely on geological, biological, deondroclimatological and other natural resource information. All of these data sets, from the soils to the pollen to the artifacts, can be altered by fire, and so our interpretations of past uses can also be altered by fire effects on these data.

Archaeological materials possess certain characteristics relevant to fire management. Evidence of the past is non-renewable. Once disturbed, modified or destroyed the information is gone forever. Spatial relationships among artifacts, features and sites are an essential component of archaeological research, and fire can disturb these relationships. A third factor is that certain sites are valued by Native Americans where specific kinds of disturbance to them or the duration of disturbance are important considerations (Cook and Rock 1994). And, last but not least, there are legal mandates requiring consideration of cultural resources in project planning and implementation that date back to the Antiquities Act of 1906.

Many thousands of sites exist in the sky island region, representing at least 12,000 years of occupation (Doyel 1993). For archaeologists, advantageous conditions of good material culture preservation and the existence of highly visible and variable sites have combined to preserve a rich cultural heritage. Although a firm chronology of occupation has not been outlined for the region, the archaeological record and early contact ethnographies indicate great variation in the technological capabilities and land use practices of previous inhabitants. Some lived in small bands and practiced hunting and gathering while others lived in larger villages and depended upon irrigation agriculture. The diversity of prehistoric adaptations and cultural remains is reflective of the natural diversity in the region today.

The known sites are particularly sensitive to fire because of two factors: many are small surface sites that appear to have little subsurface depth. Thus, our archaeological information comes primarily from surface contexts where fire impacts can be most prevalent. Even pithouse village sites, found throughout the region, tend to occur at shallow depths below the ground surface. A second factor relates to the lack of accurate chronological information for the region. Important chronometric data obtained from tree ring samples, obsidian hydration, archaeomagnetic or radio-carbon dating could be significantly altered because of fire.

CULTURAL RESOURCES AND FIRE PLANNING

Archaeologists know very little about the effects of fire on cultural resources, however agency managers must make decisions regarding treatment of these resources during managed fire situations with increasing frequency. Our information is limited primarily to analyses conducted after wild fires and a few experimental studies conducted as part of prescribed burns. During the last 20 years archaeologists have been involved in fire suppression activities to some extent. This involvement has focused on reviewing site and survey records during a fire to assist in locating fire lines and routes for heavy fire fighting

equipment, and assisting in mop up activities so that work does not disturb sites.

Fire projects were sometimes judged successful if a black line was created around a significant historic cabin so it did not burn, or if bulldozers did not demolish standing walls of a prehistoric pueblo site. We are becoming much more sophisticated in our considerations of cultural resources as archaeologists and fire managers work more closely together.

The Baker Burn in the Peloncillo Mountains along the Arizona and New Mexico border is an example of the procedures being used for the last several years by the Forest Service and other federal agencies in an attempt to protect cultural resources as part of prescribed burning (McDonald and Gillespie 1993). The archaeological survey and background research were completed in 1993 and the burn was conducted in 1994.

Forest Service guidelines regarding deliberately ignited fires include: review of records regarding cultural resource sites and traditional uses; reconnaissance-level archaeological survey to determine the likelihood of flammable properties (e.g. historic structures, wooden beams from prehistoric rooms); a complete archaeological survey of all areas of direct ground distubance such as fire equipment and personnel staging areas and fire lines; and, a complete archaeological survey of areas where fires are expected to exceed fireline intensities of 400 BTU/ft./second (Forest Service Manual 2360).

The purpose of the Baker Prescribed Burn, covering a 6,000 acre area of federal, state, and privately owned land lying just north of the International Border with Mexico in both Arizona and New Mexico, was to improve habitat quality for both wildlife and livestock and to restore a more natural mosaic of vegetation types. Burning was aimed at increasing the herbaceous plant growth by reducing the amount of woody plant cover, most notably scrub mesquites and junipers.

A cultural resources sample survey was completed. Survey was conducted on ridges and terraces where sites may be likely, and on ridge slopes where rockshelters may be present. The steep rocky mountainsides were not examined on the ground because of the low probability of past use. Staging areas received a complete survey, and survey was conducted to locate structures identified on early historic maps.

Twenty-one sites were located and recorded in this manner. Most were prehistoric artifact scatters representing seasonal hunting or resource gathering camps. One rockshelter was located as well as several historic homesteads. Potential impacts to these sites were identified as: ground disturbance from activites related to conducting the burn; damage from the fire itself; and an entirely separate issue but one that does warrant consideration as evidenced from previous burns elsewhere in the Southwest, artifact collection as a result of increased personnel in the area. The intensity of the fire was planned to be low so that burning over the prehistoric sites was not considered to result in damage. No flammable materials were noted at the sites. Black lines were placed around the historic sites so that the site areas would not burn, and the location of staging areas was modified from those originally proposed and carefully marked to avoid impacts to prehistoric sites. These measures represent a reasonable amount of caution regarding the protection of cultural resources, but we do not know for certain if they were sufficient, or perhaps unnecessary.

Our limited experience has demonstrated that we must be able to first predict, and then control, fire intensity and duration on or adjacent to cultural resources sites if we are to prevent impacts that lead to damage and loss of scientific information. In large prescribed burns, it could be necessary to exclude areas, or burn areas around sites in a very controlled manner. Burning of large areas with complex patterns of terrain, vegetation, and fuel loadings needs to be approached with considerable more caution than burning of small areas where conditions are generally more uniform (Lentz, Gaunt and Willmer, in press).

AVAILABLE DATA REGARDING EFFECTS OF FIRE ON CULTURAL RESOURCES

The first systematic study of fire effects on cultural resources in the Southwest came after the La Mesa fire in Bandelier National Monument in 1977 (Traylor et al. 1990). This wildfire burned about 15,000 acres before intensive fire suppression efforts including the use of bulldozers and miles of cat and hand lines were successful. Archaeologists were involved in the fire and conducted surveys and site inspections after the fire. Additional study conducted by the National

Park Service involved field documentation of sites, test excavations and laboratory analyses. The study was also important because fire specialists were involved to estimate fire behavior characteristics on the affected sites. Fire effects studies have more recently been conducted at Mesa Verde National Park (Eininger 1990), in the Coconino National Forest (Pilles 1984), and as part of the Yellowstone Fire study.

A study that is contributing significantly to our knowledge of fire effects on cultural resources and which can help assess whether agency protective measures are sufficient is a study conducted by the Museum of New Mexico in cooperation with the Santa Fe National Forest, Rocky Mountain Station, New Mexcio BLM, and the National Park Service. Researchers have examinied the effects of the 1991 Henry Fire in northern New Mexico on 52 prehistoric sites (Lentz, Gaunt and Willmer, in press). This fire occurred in an area that had been surveyed archaeologically so baseline data on sites were available. Examples of the kinds of impacts observed and analyzed at these sites are are summarized below (Lentz, Gaunt and Willmer, in press).

Ceramics.—Pottery is often altered as a result of fire. Results of ceramic studies from sites in the Henry Fire indicate that there were significant impacts to ceramic artifacts regardless of fire intensity. Effects included sooting, spalling, oxidation, and alteration of pigment and other physical characteristics. Some polychrome sherds were so altered through changes in color and spalling, that is the removal of surface areas of a sherd because of heat buildup, that field identification as to type was not possible. The inability to accurately identify diagnostic pottery in the field can be a major handicap since management decisions and evaluations of significance are often made based on field observations.

Ground stone.—Sooting, oxidation and spalling are common on ground stone tools. Although these effects occur from fire, the interpretive potential of tools such as manos and metates does not appear to be substantially altered. The potential loss of information comes from the elimination of palynological or macrobotanical information preserved on grinding surfaces.

Another potential information loss regarding ground stone tools is that broken ground stone tools were commonly recycled for cooking or roasting uses and are found on sites as fire-cracked rock. Depending upon the specific material type it would be difficult to distinguish fire-cracked rock, as a by-product of prehistoric domestic activities from a rock that had been cracked through exposure to more recent natural or prescribed fires. This has important implications for site and artifact interpretations.

Chipped stone artifacts.—Basic typological and functional descriptions are usually not affected by fire although there are exceptions. Core flakes may be altered through spalling and cracking to the extent that they could be mistakenly documented as angular debris. Heavy sooting and adhesions resulting from burning may limit the potential for use-wear analyses.

Chipped stone studies become more complicated relevant to fire because of heat treatment which may have occurred prehistorically. Raw lithic materials were sometimes deliberately subjected to thermal alterations, presumably to make them more amenable to reduction and creating the desired tool. Currently it is not possible to distinguish deliberately heat-treated lithic materials from lithic materials that had been modified by fire long after their use ended.

Obsidian.—Fire does affect the hydration rind that forms on obsidian tools. These hydration bands can be used to date obsidian objects, thus dates obtained can be significantly altered by fire. Archaeologists tend to place greater reliance on obsidian hydration dates when the tools have come from a subsurface context, however surface materials sometimes provide the only available chronological data when sites have no depth.

Architectural features.—Architectural features from the Henry Fire were mainly volcanic tuff used as building blocks. These blocks spalled or broke as a result of fire. We have no data on building materials such as adobe or rock cobble structures relevant to potential fire effects in sky islands.

The results of the Henry Fire indicate that fire can thermally alter all types of artifacts and that the impacts recorded were derived from the fire intensity, duration of heat, and heat penetration into the soil. Fire effects were present even on lightly burned sites, and the effects increased significantly on moderate and highly burned sites. Subsurface thermal alteration of artifacts also occurred at the moderate and high intensity fire levels (Lentz, Gaunt, Willmer, in press).

The Henry Fire also demonstrated that fuel loading is the critical variable in the severity of effects. The impact to cultural resources can be greatly reduced in prescribed fires by removing extraneous fuel loads from site surfaces prior to burning or by creating fire breaks that encircle sites. Unnatural fuel loadings, usually a result of fire suppression, lead to very hot fires with a relatively long residence time. These appear to have the greatest potential for site and artifact impacts which can easily become damage (Lentz, Gaunt, Willmer, in press). It is widely recognized that the mountain ranges of the sky islands contain an abundance of such unnatural fuel loadings.

There are additional cultural resources considerations not related specifically to the Henry Fire situation. Rock art, particularly pictographs, are especially susceptible to fire damage because they can be scorched or burned away (Pilles 1984).

A notable post fire impact is erosion. Soil erosion can occur related to fire line construction or because of the stipping of ground cover. Displacement of artifacts is the most common result. The 1994 Rattlesnake Fire in the Chiricahua Mountains is a dramatic example of potential impacts from erosion. Archaeological materials that may have been present at the upper elevations of the mountain could now be included in the erosional material that has filled Rucker Lake as a result of the fire.

Data from soil analyses indicate that soil is generally a relatively poor conductor of heat (DeBano 1988), however further experimental studies examining the depths to which heat may cause modifications to artifactual, structural and organic materials are needed.

A further consideration is that we do not know if effects on cultural materials are a result of recent fires or of past fires, including those which may have occurred during occupation or abandonment of a site. Research is necessary to determine attributes that may identify the temporal context of burning.

Analyses from the Henry Fire focused on understanding the effects of a recent fire on cultural resources. The second phase of research includes experimental situations and focuses on determining the thresholds of impacts to various kinds of archaeological materials. The ability to distinguish between fires of the past and modern fires on archaeological materials is also being investigated.

The study includes experimental burning across both real and artificial sites under different fuel loadings and within different fuel models to try to establish thresholds of fire damage to archaeological materials. Burning is initially conducted under conditions of both light and moderate fuel loading with an option to also test under heavy fuel loadings if appropriate. At both actual and artificially created sites artifacts are placed at a variety of provenience locations such as on top of the duff layer, on the ground surface but beneath the duff layer, five centimeters below the ground surface, etc.. The physical condition of these artifacts is analyzed and documented prior to placement. The actual burning is being carried out in conjunction with planned prescribed fire projects. After the burn, archaeologists recover all materials and once more analyze them to determine fire effects and assess whether or not the effects are significant in terms of loss of integrity, research and intepretive potential, or other factors.

Preliminary observations suggest that there may be important implications regarding the integrity, condition and preservation status of sites that have been exposed to controlled burns and wildfire (Lentz, Gaunt and Willmer, in press). The assumption has often been that sites have periodically burned in the past, and that low intensity prescribed burns do not create additional impacts. Data from these studies may demonstrate that there is evidence to the contrary (Lentz, Gaunt and Willmer, in press).

RECOMMENDATIONS FOR RESEARCH IN THE MADREAN ARCHIPELAGO

We have an excellent opportunity in the Madrean province to carry out fire effects studies, particularly involving interagency and cooperative efforts such as in the Malpai Borderlands project (McDonald 1994). Examples of research questions that need to be pursued include ones such as:

- How can the effects of fire of varying intensities on different artifact and feature classes be determined?
- What is the threshold temperature and fire intensity at which significant damage to cultural resources occurs?
- What is the most consistent way of predicting the effects of fire on cultural resources based on

fuel characteristics, fire characteristics, and the nature of the cultural resources?

- To what depths may heat cause modifications in artifactual, structural and organic materials?
- Do changes in the site's artifactual content brought on by fire alter the data relevant to answering the questions important to the study of prehistory that a study of an unburned site could have provided?

This last issue may be the most important because it unites all the other questions regarding fire's potential to effect cultural resources. Closely related to this is our current inability to differentiate between effects of recent burning from those of past fires. Fire history research in the sky island region (c.f. Grissino-Mayer, Baisan, Swetnam 1995) is providing a wealth of data that could be used to help assess the likelihood of previous fire impacts to cultural resources in areas where fire history records have been determined.

We also need experimental studies of fire effects on cultural resources from which to develop guidelines that can be used to more effectively and efficiently protect cultural resources in both wildfire and prescribed burn situations. The diversity of cultural resources in the sky island region, at various elevations and in a variety of topographical situations, provides an excellent opportunity to acquire data that are applicable throughout the Southwest.

Through the limited studies of the last two decades, through on-the-ground experience, and through working with fire personnel, archaeologists have learned a great deal about the potential for fire and associated activities to affect cultural resources. However, as Tom Cartledge, the Forest Archaeologist on the Santa Fe National Forest who is a lead investigator in the ongoing fire research in northern New Mexico, so clearly says on the subject:

"I believe that as we move toward management of multiple resources from an ecosystem perspective, we need to direct management efforts in our fire program, as well as in other functional areas, to those projects which will result in the greatest benefit to the greatest number of resources considered together. When we become too narrowly focused upon a single management objective, then any of the various other resource objectives tend to be viewed as constraints or limitations upon our focal objective." (Cartledge 1994). I totally agree.

Prescribed natural fire should be viewed as having long term advantages for the protection and preservation of significant cultural resources. By allowing fire to play a more natural role in the evolution of our ecosystems and by burning under more controlled conditions, impacts to our heritage resources should be less than would occur with the high intensity catastrophic fires that result from years of fire suppression and accumulated fuels. What is needed is additional experimental data to provide us better predictive potential and increased flexibility in managing fire where signficant cultural resources are present.

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Modeling Human Impacts to the Borderlands Environment from a Fire Ecology Perspective[1]

Suzanne K. Fish[2]

Abstract.--The methods and motivations for fire use varied for late prehistoric societies of the Southwest. Although fire was probably used to increase the returns from hunting and gathering on marginal lands, it seems doubtful that comprehensive burning was used as a tool within patterned agrarian settings and immediate sustaining areas. Controlled rather than comprehensive uses of fire were the key to improving natural vegetation structure.

INTRODUCTION

Current theoretical and topical orientations in archaeology address the interaction between past human populations and their environment. These themes reflect ecological frameworks that were incorporated into the social sciences by the 1950's, as exemplified by the influential publication, *Man's Role in Changing the Face of the Earth* (Thomas 1956). Cultural practices for actively manipulating resource species and ultimately modifying vegetational structure have been highlighted in recent studies of hunting and gathering societies, and have gained recognition as key processes in the transition from foraging to farming economies (e.g. Hillman and Harris 1989; Price and Gebauer 1995; Smith 1992). From a background of such interests, archaeologists are increasingly involved in examining the relationship between prehistoric peoples and pre-contact vegetation on what are now public lands of the United States.

There is, of course, no unitary means for characterizing the manner and magnitude of vegetational impact by pre-industrial societies in North America or even the Southwest. Natural vegetational dynamics over time are driven primarily by climate and secondarily by processes such as geomorphological change. However, these natural processes are more regular and subject to uniformitarian approaches than are the overlay of human interventions that affect them; no uniform reconstruction of human impact can be advanced without reference to economic orientation, population densities, or culture-specific practice. To meaningfully assess influence on a precolumbian baseline, then, it is necessary not only to specify vegetation types, but also to specify the dimensions of aboriginal population, subsistence, and settlement.

I will examine the prehistoric human fire ecology of southeastern Arizona and southwestern New Mexico from the perspective of an archaeological overview of the Malpais Borderlands, undertaken with Paul Fish on behalf of the Coronado National Forest and the Rocky Mountain Forest and Range Experiment Station. The Malpais Borderlands study area consists of approximately 1000 square miles encompassing large portions of the San Bernardino, Animas and Playas Valleys and the Peloncillo and Animas Mountains (Figure 1). Grasslands and desert shrub associations occupy the valley floors, with an elevational progression to coniferous forests on the higher peaks (Brown 1994). Despite the fact that the archaeology and culture history of the Borderlands are poorly known by comparison with other regional sectors, this area figures prominently in discussions of the impact of aboriginal burning on the structure of Southwestern vegetation.

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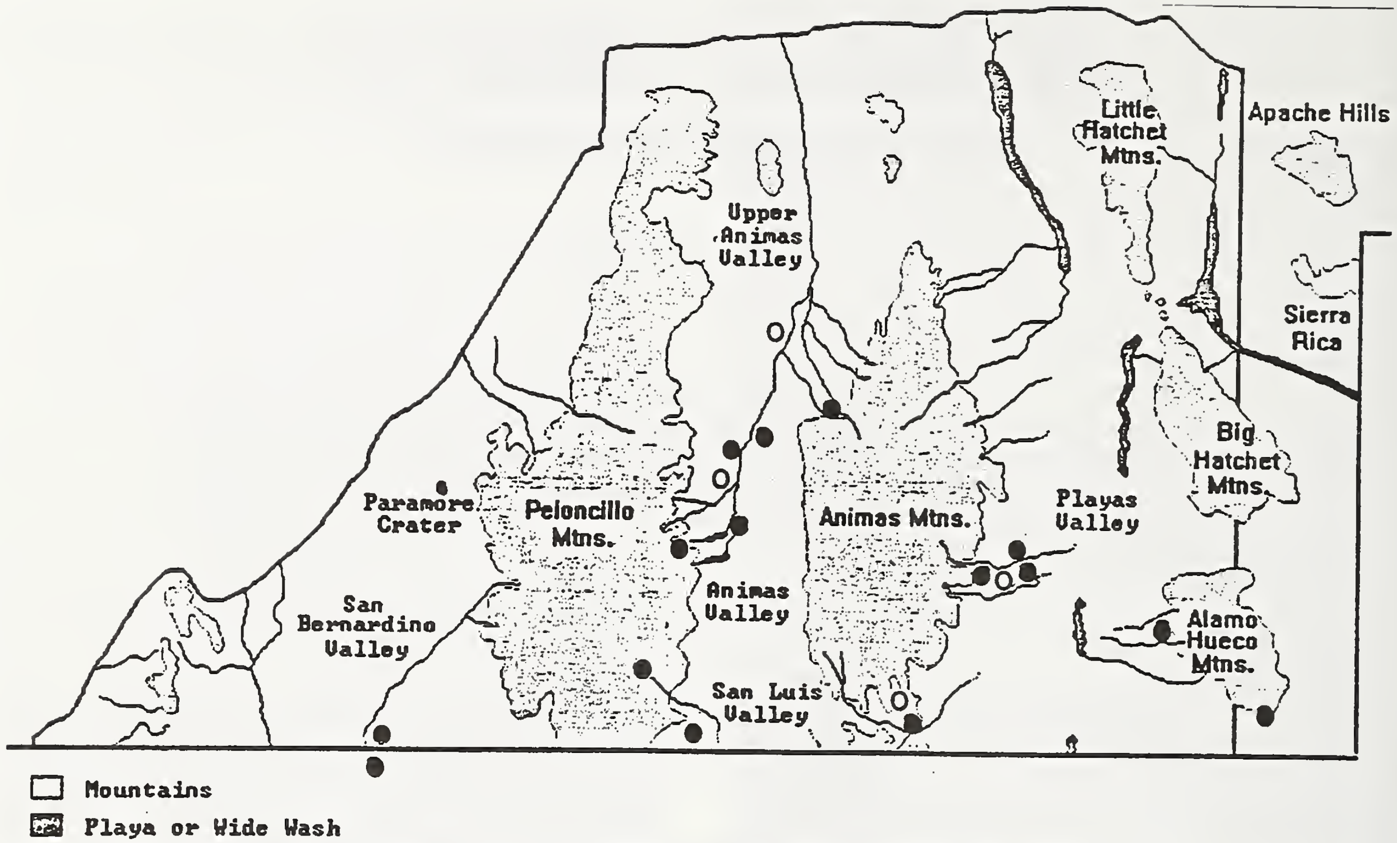

Figure 1. Distribution of large (exceeding 40 rooms), late prehistoric pueblos in the study area. ● = Pueblo O = Pueblo with known or probable ballcourt.

MAN-INDUCED FIRE IN HISTORICAL PERSPECTIVE

A decisive role for wildfire in the evolution and maintenance of desert and plains grasslands and open forests in the American Southwest has become an important component of range management philosophy and ecological thinking. According to this paradigm, periodic and widespread grassland blazes in the past encouraged vigorous return of annuals but inhibited the survival of woody plants. Likewise, cyclical wildfires removed woodland understory so that forests of large trees were open and sprinkled with clearings. Changes in fire frequencies in the wake of overgrazing and fire supression resulted in conditions that permitted the invasion of dense young trees and brush into forest understories, and shrubs and trees into grasslands by the late 1800s.

Carl Sauer (1944, 1956) and Omer Stewart (1956) are among the most influential proponents of the hypothesis that the former broad, brush-free grasslands and mature, open forests of the New World originated with repeated fires set by Native Americans as much or more than from naturally occurring conflagrations. Stewart (1956: 118) believed that the unrestricted burning of vegetation is a universal trait among "primitive" peoples. Conversely, fire suppression is limited to contemporary societies (Sauer 1956: 55); 3) Further, landscapes were repeatedly subjected to fires used in Native American hunting activities (Sauer 1944: 554, 1956; Stewart 1956: 129). Sauer and Stewart correlated this perceived universality of native burning with historic environmental change to arrive at the potent force shaping premodern grassland and forest landscapes. Some current environmental historians working in the western United States have expanded this thesis to implicate Native American burning in the overall development of North American ecosystem structure (e.g. Pyne 1982; Kay 1994).

Researchers in the Borderlands study area and adjacent regions (e.g. Cooper 1960; Branscomb 1956; Humphrey 1958, and more recently Henry Dobyns 1981: 27-44, S. J. Pyne 1982, 1984 and Conrad Bahre

1985, 1991) have assembled an array of ethnographic, archival, and newspaper accounts of historic fire in general and anthropogenic fire in particular. Conrad Bahre (1985: 190, also Bahre 1991: 124-142), reviewing local newspaper accounts of wildfires in grasslands of southeastern Arizona during the latter portion of the nineteenth century, concludes: wildfires were much larger and more frequent than at present, the occurrence of fires declined dramatically during the 1880s, cessation of large fires preceded the brush invasion of the 1890s, Native Americans (primarily Apaches) often set wildfires, and supression of fires was encouraged by Anglo settlers.

NATURAL VERSUS CULTURAL MAGNITUDES

The scale and efficacy of naturally-induced fire is a question with logical precedence in any attempt to retrodict human effects. If natural ignitions were plentiful and cultural suppression nonexistent, why would not the balance between accumulating fuels and recurrent fires be sufficiently close to preclude first order consequences of human ignitions? Under conditions approaching such a balance, why would culturally-generated fires transform or maintain vegetation types rather than accomplish only more minor and local adjustments of vegetation potential? Proponents of Amerindian burning as a primary factor in the persistence of grasslands and open forests seldom confront this fundamental issue (but see Lewis 1973 for an exemplary analysis of the interplay between aboriginal burning and natural fire regimes in California).

The thoroughness and rapidity with which naturally-ignited fires consume available fuels is particularly relevant in assessing prehistoric human impacts on Borderlands ecosystems. Bahre (1991: 124) summarizes the impressive occurrence of lightning in the study area and vicinity. Nearby portions of the Mogollon Rim country have among the highest incidences of lightning fires in the United States. Seventy-three percent of all fires since 1959 in the Coronado National Forest originated in this way. The Coronado Forest leads the Southwest in average acreage burned annually by lightning fires, despite a long history of fuel reduction through grazing. These high natural ignition rates bring the role of aboriginal burning into perspective as a residual mechanism in total fire ecology. As noted by Bahre (1991: 128), ". . .given the high incidence of lightning-caused fires, reliably documented by modern data, the relative importance of fires set by Indians is probably moot."

Although the precise relationship between natural fire and vegetation in the past is not amenable to simple or immediate resolution, it is possible to ask whether historic trends in Borderlands vegetation attributed to human agency had earlier parallels. For example, Thomas Van Devender (1995: 89-94) cites packrat midden evidence for increases of desert shrubs in southeastern Arizona grasslands about 4,000, 3,000, and 1,000 years ago, in the absence of livestock or fire suppression programs. He attributes these increases to climate, but acknowledges the unprecedented intensity and possible irreversibility of modern disturbances exacerbating this vegetational response. In any case, there is wide concurrence on the two foremost causes for a decreased scope and frequency of fire after the final decades of the nineteenth century: the removal of ground cover and fine fuels through overgrazing and the initiation of containment efforts (e.g. Bahre 1985, 1995; McPherson 1995; Pyne 1984; Van Devender 1995). Thus, it is clear that prehistoric human fire ecology of the Borderlands should be viewed in the framework of a more vigorous natural fire regime than at present.

FIRE ECOLOGY OF THE EARLY ARCHAEOLOGICAL SEQUENCE

There is a substantial body of information on the use of fire by Native Americans in the southern Southwest during the historic era (see summaries in Dobyns 1981; Bahre 1985). Because direct evidence for the burning practices of prehistoric inhabitants in the Borderlands is negligible, historic accounts and ethnographic descriptions figure prominently in modeling fire use in the past. However, analogs cannot be uncritically applied without regard for their fit to the demographic and economic profiles of prehistoric populations in particular times and places. That is to say, no single model of human fire ecology, based on historic accounts, can adequately accomodate the changing array of cultural configurations throughout the archaeological sequence. It is especially difficult to evaluate human influence in the distant past when both vegetation and culture were quite different from their counterparts of later times.

Little is known about the earliest Paleoindian hunters of the study area beyond the existence of kill and butchering sites for large, extinct game such as mammoth. Paleoindian patterns of residence and seasonal mobility are uncertain, as are all additional aspects of subsistence. Vegetation assemblages from packrat middens in the uplands suggest that grasslands emerged from more arboreal post-Pleistocene vegetation types about 8,500 years ago (Van Devender 1995: 89), but pollen diagrams from still earlier Paleoindian kill sites on valley floors also conform to grassland patterns (Mehringer and Haynes 1965: 22; Martin 1963a,b). The use of fire drives in the hunting of extinct big game is plausible but highly speculative as to overall effect.

Forms and distributions of desert grassland and scrub resembling those of the present were in place by approximately 4,000 years ago (Van Devender 1995: 89). With the demise of large Pleistocene fauna, Borderlands occupants shifted their subsistence focus shifted to a broader set of smaller animals and gathered diverse plant resources, to judge by plentiful grinding stones and other implements in artifact assemblages. Again, few details are known about the lifestyles of the Archaic peoples who scceeded Paleoindian hunters, but the discovery of occasional structures in the later centuries before agriculture implies intervals of relatively prolonged occupation in some locales (Fish and Fish 1994).

In a cross-cultural survey of worldwide hunting and gathering societies, Lawrence Keeley (1995) found a correlation between the use of fire for landscape management, dietary reliance on seeds and nuts, intensive manipulation of wild plants, and decreasing mobility. Indeed, this combination of attributes fits those historic cultures of the western United States for whom the most systematic and spatially complex burning practices for plant management have been recorded, such as the Shoshone and Paiute of the Great Basin (Steward 1938; Stewart 1939) and a variety of groups in California (e.g. Lewis 1973; Bean and Saubel 1972; Anderson 1993). With longer residence time in single locations, management efforts are both increasingly practicable and necessary to obtain food supplies from territories of limited size. Intensive foragers of the Archaic Period just before the transition to the still more intensive land use of farming may well have been the premier vegetation burners of Borderlands prehistory.

In the Tucson Basin and on its eastern borders, the agricultural threshold now can be confidently pushed back to 1,000 B.C. and probably several centuries earlier (B. Huckell 1996; Mabry 1996; S. Fish, Fish and Madsen 1989). The recovery of corn pollen by Owen Davis (1994: 22) from the Animas Creek Cienega near the Gray Ranch Headquarters, in a level dated earlier than 3000 B.P., suggests that the appearance of farming was generally synchronous in the Borderlands. Where Late Archaic subsistence remains have been studied near Tucson, even the earliest cultivators possessed a full complement of aboriginal food crops, and corn remains are as common in their pithouse villages as in later agricultural settlements (L. Huckell 1996a,b).

FIRE ECOLOGY OF LATE PREHISTORIC AGRICULTURALISTS

Developments among the earlier pottery-making agriculturalists can be sketched only in broad outline for the Borderlands. More information is available for later prehistoric times, when preliminary distributions can be compiled for the largest residential sites ever occupied in the study area (Figure 1). These sites date to the last centuries before Old World intrusion, from about A.D. 1250 to 1450. Because archaeologists have undertaken little systematic inventory or excavation, discussion of human fire ecology during this era draws upon more numerous and refined studies to the west and north in southern Arizona, particularly those of the Hohokam. The selection of ethnographic analogs similarly emphasizes agricultural groups in those areas.

Disjuncture with Apache Practices

At this point it is useful to contrast the cultural contexts of fire use in Borderlands history and prehistory. Both proponents and discounters of aboriginal burning as a primary force in vegetational structure invoke accounts of Apaches and similar groups between the time of initial Spanish encounters and the late nineteenth century. For a variety of reasons, it is improbable that these observations are valid reflections of pre-contact times. In addition to their tenuous cultural continuity with late prehistoric agriculturalists of the Borderlands, the Apache followed lifestyles that were shaped by uniquely historic factors.

Thoughout the periods of Spanish, Mexican, and United States hegemony, the Apache were simulta-

neously in conflict and economically intertwined with these dominant societies and neighboring indigenous farmers. Mobility was heightened by ongoing hostilities, capabilities for animal transport, and heavy economic involvement in raiding and trading. Although Apaches gathered plant resources and sometimes cultivated for subsistence needs, opportunities for extended residence and intensive forms of land use, and thus their overall dependence on the outcomes of such practices, were limited. Furthermore, some Apache had more than a passing involvement with livestock. For example, Henry Dobyns (1981: 24) cites a description of Apache livestock along Pinal Creek by 1830. Livestock obtained in perpetual raiding also had to be temporarily sustained, even if later disposed of through trade.

Apache patterns of land use, including the application of fire, could be expected to diverge from those of relatively intensive or sedentary foragers as well as those of late prehistoric agriculturalists. Mobile (and to some extent unpredictable) Apache lifestyles, in conjunction with low populations, would hold few incentives for regular, intensive, and fine-tuned manupulations of the environment. Relocation was a ready option and sometimes an urgent necessity. It is not surprising that historic records of Apache burning, even though inflated by automatic attributions, are repeatedly linked with situations of conflict (Bahre 1985; Dobyns 1981) or that fire chronologies derived from Borderlands tree ring studies suggest a correlation between increased frequencies and intervals of heightened warfare (Kaib 1996).

Late Prehistoric Settlement Patterns

After A.D. 1200, the largest class of residential sites in Borderlands settlement patterns are characterized by pueblo-style adobe structures ranging from about 40 to perhaps several hundred contiguous rooms. Compounds, consisting of more dispersed sets of rooms surrounded by a wall, were constructed at moderate-sized sites near the western boundary of the study area; room numbers are more difficult to estimate with this type of architecture. Locations of the large settlements correspond to better situations for irrigated and floodwater farming along mid-valley stretches of the largest primary drainages and along mountain edges where tributaries with large upland watersheds emerge (Fig. 1). The importance of agriculture is inferred from these hydrological settings, but abundant corn remains have been recovered in each of four substantial excavations at large sites (Kidder, Cosgrove, and Cosgrove 1949; McCluney 1965a,b), and in a number of dry caves (Lambert and Ambler 1961).

Small residential settlements and other site types are numerous throughout the study area. Occurrences are so spottily reported, however, that their spatial relationship to the larger sites in more inclusive patterns cannot be documented. The presence of probable Casas Grandes-style ballcourts at four large sites in the study area (Figure 1) and in three just beyond its edges suggests that each of these sites served as a center for surrounding smaller settlements without this kind of communal architecture. In the area of Hohokam culture (S. Fish and Fish 1994) and in other parts of the Southwest (P. Fish and Fish 1994; Adler 1996), sets of closely interrelated outlying settlements which share a relatively large center are termed "communities". The central site contains the community's buildings for public events and ritual such as ballcourts, platform mounds, or big houses, depending on the prehistoric culture. The roughly regular spacing of large Borderland sites (Figure 1) resembles the spacing of centers in other community distributions in the Southwest (Fish and Fish 1994; Fish 1996), and the presence of additional ballcourts may be revealed by future investigations.

A ZONAL MODEL OF FIRE ECOLOGY

Just as different economic orientations have distinctive implications for burning as a management tool, spatial variability in burning practices should be expected within a single subsistence system. I will use the zonal layout of territory in communities as the principal point of reference in modeling locational variability in fire use for late prehistoric agricultural societies. Community boundaries encompassed the primary sustaining area for the entire set of member settlements. Among the neighboring Hohokam, community territories exhibit an internal zonal arrangement corresponding to differing topographic opportunities for cultivation and the procurement of wild resources (Crown 1987; Fish and Fish 1989, 1994b; Masse 1989). Thus, fire would have been differentially applied across the zones of a community and in the land beyond its boundaries according to the manner of land use in each locale.

Room numbers have been estimated for seven of the better studied large pueblo-style sites (Table 1). These numbers provide a basis for roughly calculating population size and thereby inferring the corresponding requirements for irrigated land and firewood. Similar estimates can not be derived for the remainder of community residents in smaller associated settlements. Even as minimal estimates, however, the figures for large sites aid in illustrating the relative magnitudes of land use categories and their implications for anthropogenic fire.

Zones of Intensive Cultivation

Communities usually include a highly productive core area of hydrologically favored agricultural land. Such a core in the study area would consist of land irrigated by small networks of canals from primary or large secondary drainages. At the modest scale of Borderland watercourses, formal irrigation would intergrade with extensive floodwater systems, in which long ditches serve a series of fields (cf. Doolittle 1984; Nabhan 1986; Withers 1973).

Within the radius of intensively cultivated land, uncontrolled fire would have posed an unwelcome threat. Nearby village structures with highly flammable roofs contained household possessions and, more importantly, internal or adjacent facilities held food stores. During the growing season, crops in fields would be threatened by fire. Throughout the year, however, fire might damage other vegetational elements of intensively farmed locales. As in Pima agriculture (Rea 1981, 1983), field edges and hedgerows may have consisted of woody plants that were selectively spared in clearing and herbaceous species of disturbed ground. Together, these taxa furnished edible resources and firewood and harbored a concentrated supply of small game. Supported by seepage, species such as mesquite grew along outer edges of canals. A mesic flora also lined canal banks, extending riparian resources into the zone of cultivation. In adjacent northern Sonora, traditional farmers still plant "living fencerows" of cottonwood and willow to stabilize field edges on floodplains and to safely deflect nutrient-rich flood flows over fields (Sheridan and Nabhan 1978).

Spatially discrete uses of fire in intensively farmed zones are clearly documented for aboriginal and traditional farmers of the southern Southwest. Farmers sometimes burned piled brush in the initial clearing of fields (Castetter and Bell 1942: 125) and thereafter possibly removed weedy growth on fallowed land. Canal channels were similarly cleaned at times. Burning to clear water delivery systems may have been extended to the dense growth of cienegas and drainage bottoms in locations where canals and

Table 1. Population and land use characteristics for selected large, late prehistoric pueblos in the borderlands study area.

Site #	Site Name	No. of Rooms[1]	Estimated Population[2]	Hectares of Farmland[3]	Cords of Fuelwood[4]
LA 1369	Pendleton	125	125	50	250
LA 4980	Box Canyon	175	175	70	350
LA11823	Joyce Well	200	200	80	400
LA31050	Culberson	225	225	90	550
LA54033	Double Adobe	200	200	80	400
LA54034	Cloverdale Park	75	75	30	150
LA54038	Timberlake	200+	200	80	400

[1] *Number of rooms is based on National Register of Historic Places nomination forms for the Animas District.*

[2] *Estimated population is derived from Longacre's (1970) room to person ratio.*

[3] *Hectares of irrigated land are estimated using Castetter and Bell's (1942: 54) ratio of persons to farmland among the Gila River Pima.*

[4] *Following Ravesloot and Spoerl (1994), per capita fuelwood consumption is based on a modification of Plog's (1992) estimate for prehistoric Mogollon Rim populations. Requirements for Borderlands inhabitants are reduced by 25% because of milder climate.*

ditches headed; an additional goal might have been to enhance the productivity of non-woody riparian resources (Rea et al. 1983; Gary Nabhan, personal communication). Such practices could account for the charcoal of riparian species recovered from Borderland cienegas in levels predating the late nineteenth century (Davis 1994). In all of these cases, unrestrained fires would have been destructive in residential and agricultural environs.

Amounts of irrigated land needed to supply projected populations at seven large Borderlands sites are estimated in Table 1. Quantities are based on historic Piman farming, in which at least 2.5 persons could be supported by one hectare of irrigated land (Castetter and Bell 1942: 54). Although irrigable land is intermittently distributed along adjacent streams for all but one of these sites, in no case does it to satisfy even half of the projected acreage. This lack of congruence suggests a major role for alternative methods of agricultural production and gathering.

Areas of Runoff Cultivation

Somewhat removed from well-watered community sectors, more marginal plantings were situated on ephemeral tributaries. Floodwater plots of secondary quality probably were interspersed among more extensive fields watered soley by the concentration of surface runoff. Impressive arrays of stone devices for runoff diversion and containment are widespread in portions of southern Arizona and northern Chihuahua that surround the Borderlands. For example, in three Hohokam communities of the Tucson Basin, simple cobble mulches called "rockpiles" and small contour terraces cover between 550 and 600 hectares. In the two reliably-bounded examples of these large territorial units (136 and 146 sq. km., respectively), such fields constitute about 2% of community land (Fish and Fish 1994). Low terraces and checkdams are prominent stone features of the Casas Grandes region of Chihuahua (Di Peso 1984; Minnis and Whalen 1992)), and miles of large stone grids cover the upper terraces of the Gila River near Safford, Arizona (Masse 1989).

Historically, aboriginal and traditional farmers most commonly constructed runoff diversion devices of earth and brush. These materials would seldom be preserved in the archaeological record. Stone features, then, represent a minimal indication of prehistoric use. The lack of reports of runoff features in the Borderlands may reflect the rudimentary level of archaeological inventory or a predominance of earth and brush materials.

If present in the study area, as seems likely, zones of runoff farming define a further circumscription of anthropogenic fire. Agave has been documented as a principal cultigen in many Hohokam runoff field complexes and cultivated cacti have been suggested (Fish and Nabhan 1991). Such drought-adapted plantings are strong probabilities in the arid Borderlands as well. These are perennial crops, leaving no off-season in which burning of weeds could be freely implemented. Significantly, succulents tend to be among the least fire-tolerant of desert plants (McLaughlin and Bowers 1982; McPherson 1995: 137-140; Cave and Patten 1984).

Variable Radii of Gathering

The disturbed environs of prehistoric cultivation provided secondary resources in weedy plants (Fish and Nabhan 1991). Typically, these are annuals that mature rapidly and produce edible greens and abundant seeds. Intensive gatherers burn plots to increase the production of such species, but extensive firings by agriculturalists would be relegated to zones beyond both intensively and extensively cultivated land. The lower needs of farmers than foragers for weedy harvests may have been largely satisfied from field margins and fallow land.

Labor expended in agriculture can be seen as a counterbalance to investments of time in wild plant manipulation near and beyond community boundaries. It seems doubtful, for instance, that irrigation and floodwater farmers would have had sufficient time or incentive to routinely burn meadows in high elevation forests, despite potential enhancement of edible species. The most intensive manipulations of wild plants by agriculturalists might be expected near the margins of cultivation rather than at greater distances. Moreover, practices undoubtedly reflected the fact that wild resources respond variably to fire. Grasses and other seed bearing annuals yield more abundantly after burning, for example, while fruit-bearing cacti are damaged.

Hunting Strategies

Vegetational effects of anthropogenic fire at the broadest scale are often linked to hunting (e.g. Stewart

1956, Kay 1994). Indeed, fire was commonly employed for this purpose among cultures of the southern Southwest and northern Mexico (Dobyns 1981: 42). Hunters lit fires to flush out their prey, to direct or contain the flight of animals, and to promote plentiful, palatable fodder. Mounted hunters in post-contact times could rapidly pursue fleeing game; prehistoric hunters on foot had more compelling reasons to carefully position fires so as to constrain the path of flight. Practices designed to simultaneous control fire and animals are illustrated in a description cited by Dobyns(1981: 43). Yaqui hunters set a fire in a circle when there was no wind so that it burned toward the center. Men waited at the perimeter for game escaping outward. Even though such means could be used to exert control, the Pima selected areas more distant from villages and fields for fire drives than for other forms of communal hunting (Rea 1979: 114-115).

Burning at a landscape scale, such as comprehensively throughout the grasslands on Borderlands valley floors, is conceivable as a strategy to improve the carrying capacity for herbivores (cf. Dobyns 1981: 39). In light of the already high incidence of natural fires, such a program would seem to entail persistent effort for an uncertain level of added benefit, while posing risk to villages and fields in predominantly lower valley situations. When descriptions of aboriginal burning for forage improvement provide detail, areally discrete and less ambitious goals are often expressed. Burning of more restricted extent, such as near a water source or in a forest clearing, offers the distinct advantage of attracting and concentrating game in a precise, predictable location for hunters.

Radii of Fuel Supplies

During any prolonged occupations of large sites, residents had continuing need of domestic fuel. Cleared land and culturally modified vegetation in the vicinity of settlements added to the radius of firewood supply. Lower elevation woodlands of adjacent mountains must have been important sources. Fred Plog (1992) estimated 2.7 cords of firewood per capita annually for prehistoric populations of the Mogollon Rim. Adjusted downward to two cords for the somewhat milder Borderlands climate, areas of sustainable yield in low density emory oak woodland (Touchan 1988) range from 6.5 to 23 square miles for projected village populations in Table 1. Repetitive harvesting of deadwood may have affected natural fire frequencies in areas of longterm fuel gathering, and it seems probable that intentional burning would have been avoided to conserve nearby supplies.

FIRE ECOLOGY OF ANTHROPOGENIC LANDSCAPES

I have reviewed evidence suggesting that the methods and motivations for fire use were spatially differentiated for late prehistoric societies of the Borderlands. The potential for small scale irrigation and intensive floodwater techniques is modest when compared to the scale of agriculture in other parts of the southern Southwest. In addition to runoff cultivation of marginal land, management of natural vegetation to increase the returns of hunting and gathering are likely to have included the localized use of fire. It seems doubtful, however, that comprehensive burning was implemented within zonally patterned agrarian settings and immediate sustaining areas for wild resources.

The greatest variety of resources can be obtained from heterogenous rather than homogenous landscapes. Mobile populations could have attained such variety by periodically moving from place to place, an option not equally viable for farmers. Vegetational mosaics would have been most valuable to agriculturalists for gathering as well as for hunting.

High incidences of lightning in the Borderlands guaranteed ready access to resources that proliferate under fire succession. Cultural practices could have accelerated vegetational response by introducing fire in seasons of low natural ignition and by increasing frequencies to the extent that fuel buildups allowed (Lewis 1973; McPherson 1995). Nevertheless, controlled rather than comprehensive uses of fire were the key to achieving a diversified and resource-rich vegetation that improved upon natural structure.

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Fire Effects on Aquatic Habitats and Biota In Madrean-type Ecosystems: Southwestern United States

John N. Rinne and Daniel G. Neary[1]

Abstract.—Intense wildfires effectively remove vegetation, degrade watershed condition, and result in altered stream hydrographs and increased sediment input to streams. Case histories from five headwater streams in Arizona and New Mexico show effects of wildfire on aquatic habitats, fishes, and their food supply may be marked and long-lasting. Recent (1989-90) wildfires effectively extirpated two populations of brook trout (*Salvelinus fontinalis*), one of rainbow trout (*Oncorhynchus mykiss*) and the type locality population of the endangered Gila trout, (*O. gilae*). Aquatic macroinvertebrates densities declined to zero within 1 month after the fire, and diversities dropped 25 to 70% a year later. Trout populations re-introduced 1 year after the fire declined 85 to 97% in a two-year period. Highly variable streamflow and fluctuations in Gila trout populations in one stream in southwestern New Mexico suggest the effects of wildfire on stream habitats and biota may be long lasting. Forest Service management and research must continue to cooperate in examining the effects of not only wildfire but prescribed fire on aquatic habitats and biota within southwestern riparian areas.

INTRODUCTION

In the Madrean Floristic Province of the southwestern USA and northern Mexico, there is a direct correlation between surface elevation and precipitation (Dunbier 1968) and ultimately the occurrence of perennial streams. The mountain ranges of the Basin and Range Geomorphic Province are more mesic than are the desert floors (or basins) because of orographically produced precipitation. These montane "sky island" landscapes thus contain the major proportion of the perennial streams in the area (Rinne 1995). However, the vagaries of the local climate (Green and Sellers 1964) result in periodic low and intermittent flows, even in these upper elevation streams.

In the streams of the more mesic montane areas, salmonids occur naturally (Rinne 1985, 1988) or have been widely introduced for sport fishing. The Apache trout (*Oncorhynchus apache*) and the Gila trout (*O. gilae*) evolved in montane streams in Arizona (Miller 1972) and New Mexico (Miller 1950), respectively. However, introductions of nonnative salmonids, principally rainbow (*O. mykiss*), brown (*Salmo trutta*), and brook (*Salvelinus fontinalis*) trout for sport fishing have significantly reduced both the historic range and numbers of both Apache and Gila trout (Rinne 1985, Rinne and Minckley 1985).

In addition to fish introductions, man-induced land management activities (Debano and Schmidt 1989), instream enhancement efforts (Rinne 1981b; 1982), water harvesting (Rinne and Minckley 1985), and natural disturbance events such as drought, wildfire, and flooding (Rinne 1981a, Rinne and Lafayette 1991) have collectively impacted these native trouts through habitat alteration and loss. Because of variability of streamflow and land management activities, both native and introduced trouts now live in marginal habitats.

Historically, wildfire has been a natural disturbance event on the watersheds of Madrean montane

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regions. These fires occurred every 4-5 years (Swetman 1990), and were ground level and understory in nature (Dieterich and Hibbert 1988, Wright 1990). Accordingly, forest vegetation, largely ponderosa pine (*Pinus ponderosa*), occurred in more open stands, and understory was reduced (Covington and Moore 1992).

Since the beginning of the 20th Century, suppression and control of wildfire have altered the natural process of periodic burning, and have resulted in increased fuel loads, understory and brush, and stand density (Wright 1990, Covington and Moore 1992). In combination, these factors can produce intense, rapidly spreading wildfires that consume vegetation and ground litter over vast portions of forest watersheds.

In the Southwestern USA and northern Mexico, the fire season of May-June is immediately followed by the summer monsoon season in July-August. Accordingly, the denuding of watersheds by wildfire is potentially followed by localized, heavy precipitation and runoff into streams (Rinne and Medina 1995). To date, there is no information on the effects of wildfires on the quality and quantity of these runoff events on southwestern riparian-stream ecosystems. Understanding these effects on aquatic resources is basic to effective and responsible management of National Forest lands.

The recent wildfires in Yellowstone National Park brought into focus the issue that we cannot control all fires, and some will destroy resources in spite of all efforts to control them. The Yellowstone fires have a very long fire return interval (200-300 years; Romme and Despain 1989) compared with semi-arid forest ecosystems of the Madrean Archipelago, where historically (pre-suppression era) fires in ponderosa pine burn at 3-5 year intervals (Dieterich and Hibbert 1988, Swetnam 1990). The former have a greater potential for destruction because of fuel accumulations over centuries. In the Southwest, fuel accumulations in ponderosa pine between 40-50 years would be roughly equivalent to those of the Yellowstone region. Nevertheless, the effects and use of fire in the two ecosystems are very different.

Anderson et al. 1976, Tiedemann et al. 1979, and Lyon et al. 1978 found few specific studies of the effects of fire on fishes via either post-fire changes in water quality or quantity. Gottfried and Debano (1990) reported that, although nutrient levels changed significantly in a stream following a prescribed fire in ponderosa pine in east-central Arizona, changes were too small to adversely affect water quality. Most water quality studies have not addressed the potential effects of short-term nutrient enrichment of the riparian zone and water on the complex aquatic faunal assemblages. The timing of nutrient and sediment discharge, for example, could adversely affect reproduction of aquatic fauna. On the other hand, nutrient enrichment of the aquatic environment could be highly beneficial to algaes and aquatic macrophytes that flourish under such conditions.

Prescribed fire has been widely used as a tool to manage fuel and vegetation on southwestern watersheds (Dieterich 1980, Harrington and Sackett 1988, Krammes 1990). Since the 1970s, prescribed fire has been used in an attempt to reveal and at the same time mimic what has occurred naturally in fire ecology for centuries (Covington and Moore 1992). Every year, thousands of hectares are burned by prescription in the U.S. Forest Service's Southwest Region. There is an urgent need to address both natural wildfires and those prescribed by man, and their comparative impact on management of all natural resources on forest lands.

This paper reviews primarily the effects of wildfires on riparian-aquatic habitats, and discusses three case histories of wildfires in Arizona and New Mexico. Specifically, we:

1. Review and discuss information on altered stream hydrographs and sediment production from watersheds affected by wildfires;
2. Present data on fish and macroinvertebrate populations following a 12,000 ha wildfire that occurred in summer 1990 in Arizona;
3. Review the aftermath of a 4,000 ha wildfire that occurred in summer of 1989 in New Mexico and its effects on an endangered trout; and
4. Suggest the possible influences of a wildfire 40 years ago on another stream containing an endangered trout.

Our objectives are to:

1. Increase the level of awareness of the potential effects of wildfire management on stream habitats, fishes, and fisheries; and
2. Make a plea for the need to monitor and conduct additional research on the potential effects of fire in the Madrean Province on fish habitats and populations.

This is not an exhaustive but rather and introductory document on the effects of both prescribed and wildfire on fish and their habitats.

WILDFIRE EFFECTS

Water Yield

Water yield increases from prescribed fires and wildfires in Madrean-type ecosystems are discussed in more detail in DeBano and Neary (this volume). Apart from immediate increases in surface runoff which can transport ash and sediment into streams, a major concern is that if more precipitation continues to leave a watershed as surface runoff, baseflows will eventually decrease. In extreme conditions, perennial streams could become ephemeral, with devastating effects on aquatic ecosystems.

Peakflows

The effects of fire on storm peakflows are highly variable and complex (DeBano and Neary). The high velocities and flow volumes of peakflows are responsible for most sediment transport, and alteration of channel geomorphic characteristics that adversely impact fishes. Watersheds in the Madrean Province are much more prone to enormous peakflow responses due to geomorphology (high elevation ranges), climate (monsoon weather conditions and a close source of moist, tropical air), and soils (shallow, clay-rich, and potentially hydrophobic). Peakflow increases of 3 - 971 times have been measured after wildfires in the northern reaches of the Madrean Province (DeBano and Neary).

Sediment

Sediment Yields From Fires

In North America, the regions where wildfire accounts for the highest portion of total sediment yield include the Madrean Archipelago and the chaparral steeplands of southern California (Swanson 1981). Natural sediment yields can range from 0.001 to 5.530 Mg/ha. Sediment yields usually are the highest the first year following the fire and decline in subsequent years as vegetation recovers. However, this process is slower in the Madrean Province so recovery times can be much longer than in more humid regions. All fires increase sediment yield, but wildfires produce the largest amounts (28 - 369 Mg/ha).

Debris Flows

Debris flows are the largest, most dramatic form of mass wasting to deliver sediment to streams. They can range from slow moving ash slurries and earth flows to rapid avalanches of soil, rock, and woody debris. DeBano and Neary discuss this topic in more detail. Debris flows can account for 50% of the total post-fire sediment yield delivered to stream channels in some ecosystems.

The impacts of debris flows on riparian ecosystems are multiple, and largely negative. A stable stream channel reflects a dynamic equilibrium between incoming and outgoing sediment and streamflow. Large, sudden inputs of sediments into streams cause rapid aggradation of channels. Although increased peakflows after fires can also produce channel downcutting (degradation), the process of aggradation from debris flows dominates in the short-term. Narrow, distinct channels turn into broad, braided systems. The results are damaged fish habitat, reduced macroinvertebrate populations and diversity, and extirpation of fish populations.

Water Chemistry

Undisturbed ecosystems usually have tight cycles for major cations and anions, resulting in low concentrations in streams. Disturbances such as fires that interrupt normal nutrient cycling processes can result in the increased concentrations of inorganic ions in soil solution and leaching to streams via subsurface flow. Inputs of nutrients to streams also can increase during post-fire washoff of ash or during aerial drops of fire retardant.

Excess nutrients carried to streams can increase aquatic plant productivity, reduce the potability of water supplies, and produce toxic effects in aquatic organisms. Anions such as phosphate and cations such as calcium and potassium can be exported from watersheds at 10 times their normal rate immediately after severe disturbances, but don't necessarily alter water quality as far as fish are concerned. During stormflows after the Dude Fire (discussed in the next Section on Case Histories), there was some initial concern that the anion-cation content of the ash slurry may have had a toxic effect on salmonids.

However, aside from the limited observational data on trout mortality following the Dude Fire (see below) there are no quantitative data to unequivocally support this hypothesis, and the elevated suspended sediment concentrations were certainly enough to produce a purely physical effect.

Studies of water chemistry after wildfires show increases in NO_3-N concentrations with maximums in Madrean ecosystems ranging from 0.6 to 12.0 mg/L. Increases in NO_3-N would be higher if volatilization into the atmosphere were not a major pathway of nitrogen loss. Except for the municipal water quality implications of elevated NO_3-N concentrations, post-fire water changes in Madrean Province ecosystems have generally not been demonstrated to be a problem for fishes. However, ammonium ions (NH_4^+) released by retardants dropped during wildfire suppression can be a toxicity problem for fish and macroinvertebrates. Retardant loads dropped parallel to and over streams can kill fish for distances in excess of 1000 m.

Temperature

Large fires can raise the temperature of especially small headwater streams during the conflagration. However, more serious and long-term problems arise due to direct heating of the water surface by the sun because vegetational shading to the stream has been destroyed. Increases up to 17 degrees Celsius have been measured . Rising water temperatures reduce dissolved oxygen (O_2) concentrations; O_2 concentrations <10 mg/L create problems for salmonid survival. Increases of 1-5 degrees C that are not a problem at sea level could become problematic at high altitudes of the Madrean Province. However, effects of fire on stream temperatures relative to fish have not been studied in the Madrean system.

CASE HISTORIES IN SOUTHWEST

The Dude Fire, Arizona

In late June 1990, a lightning strike set off a wildfire that consumed over 12,000 ha of forested land below the Mogollon Rim on the Tonto National Forest of central Arizona (Fig. 1). It destroyed over 50 homes, cost several million dollars for suppression, and resulted in the loss of 6 lives. The Dude Fire is now known as both the largest and the worst wildfire in Arizona history in terms of hectares of forest burned and loss of human life and property.

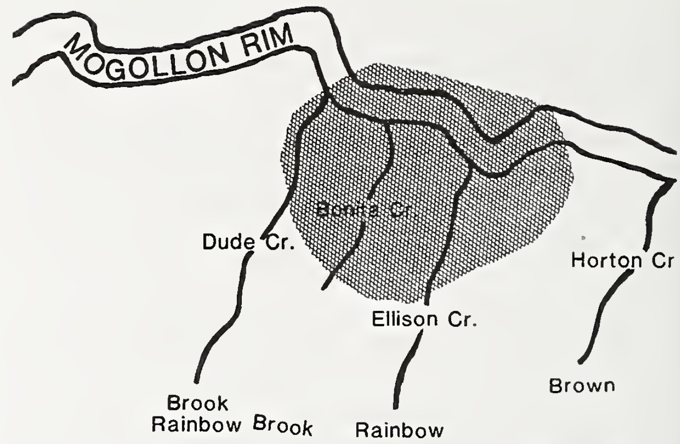

Figure 1. The relative size and positioning of the Dude Fire across the watersheds of Dude, Bonita, and Ellison Creeks, below the Mogollon Rim.

The Dude Fire burned across the watersheds of Dude, Bonita, and Ellison Creeks (Fig. 1). These streams had been under study since 1985. Although a tragedy in many ways, this natural disturbance event provided an excellent opportunity to examine the effects of wildfire on wild salmonid populations. In 1985, we began collecting baseline information on water quality, fish and macroinvertebrate populations on a group of contiguous first-order, headwater streams below the Mogollon Rim. These data (Rinne and Medina 1988, Rinne 1990), designed to serve as a "frame of reference" for information to be collected following proposed changes in grazing strategy on several allotments (Rinne and Lafayette 1991), fortuitously provided a basis for evaluating the effects of post-fire events.

Postfire hydrology

The first runoff events resulting from the developing summer monsoon season occurred between July 6th and 10th. Because of the small nature (volume) of these flow events and the availability of an extensive (5-10 cm) ash layer in both the riparian corridor and immediate hillslopes, these flows were described as "slurry flows." Concentrations of up to 700,000 mg/L suspended solids were recorded from grab samples in headwater reaches of these streams, where base flows approximate 10.6 m^3/min). With continued summer thunderstorms, larger flood events began on July 10-11 and persisted intermittently for 2 weeks.

Effects on resident fishes

Prior to the fire, trout densities in the three streams ranged between 360 and 440 fish/km (Rinne and Medina 1988; Table 1). Sampling immediately postfire (7-2-90) and prior to runoff events indicated that trout populations were slightly, although not significantly ($P > 0.05$), reduced. Although fish were not sampled immediately after the "slurry" runoff events, we observed fish swimming in all three creeks. A few dead trout were observed in pools or on streambanks.

On July 25th after subsidence of flooding no trout were captured in the reach of stream in which original stock was introduced (Table 1). Sampling after the summer monsoon season in the same reaches in October, 1990 again yield no trout. In February 1991, the entire reaches of Dude, Bonita, and Ellison creeks affected by the fire were sampled, and a single large (300 mm) brook trout was taken about 100 m from the head spring of Dude Creek. Thus, the flooding following the Dude Fire had effectively extirpated salmonids in these headwater streams below the Mogollon Rim.

Table 1. Comparison of the effects of fire on trout densities (estimated mean number per km) in three streams below the Mogollon Rim, central Arizona.

Stream	Pre-fire	7-2-90	7-25-90	10-90	2-91
Dude Creek	360	260	0	0	1
Bonita Creek	440	360	0	0	0
Ellison Creek	400	160	0	0	0

Fish restoration activities

In May 1991, 11 months after the fire, rainbow trout were restocked in Ellison Creek and brook trout in Bonita Creek (Table 2). The objectives of this effort were to determine:

1. If fish would survive atypical post-fire flood events without extensive, presumably toxic ash flows;
2. How surviving populations might respond in numbers and displacement;
3. If reproduction would occur; and
4. How the two species of fish would respond through time to changes in stream habitats and a potentially altered aquatic macroinvertebrate food base (Table 3).

Table 2. Response of stocked rainbow and brook trout to post-fire hydrology, May 1991 to July 1993. Numbers are total individuals either stocked (May 1991) or recaptured for the two streams.

Stream	May 1991	October 1991	June 1993	July 1993
Bonita	85	22	23*	34
Ellison	58	23	5	2**

35 young-of-year taken

*** 50+ young-of-year taken*

Sampling in October 1991 after a significant monsoon season revealed introduced fish stocks were reduced by over 60% in both creeks. Of trout recovered, none was over 0.5 km downstream from introduction sites, and no individuals were collected upstream of stocking sites. Fish were in excellent condition in Bonita Creek and individual growth from May to October was very good (mean Total Length (TL) 162 mm vs mean TL of 200 mm, respectively). No young-of-year rainbow trout were taken in Ellison Creek, indicating either the lack of successful spawning following introduction, or loss of young-of-year through summer flow events.

Because brook trout spawn in autumn, Bonita Creek was sampled in June 1992 following winter flow events. This species not only maintained numbers equal to October 1991 (Table 2), but, mean size had increased from 200 to 240 mm TL, spawning had been successful and fry were of a larger mean size compared to pre-fire (mean 80-90 mm TL vs mean 50-

Table 3. Aquatic macroinvertebrates per square ft surber sampler, in streams prior to and following the Dudefire.

Stream	Pre-fire 1985-86	Immed. post-fire July 3	Post-sludge flow July 13	Post-flood July 26	Post-winter May 91
Bonita	500-800	350	No data	8	553
Dude	500	460	No data	12	259
Ellison	700-900	1,000	100	0	608

Stream	Oct 1991	June 1992	November 1992
Bonita	4,400	3,800	1,100
Dude	2,800	800	3,200
Ellison	6,500	2,900	1,300

60 mm TL, respectively). Accordingly, the brook trout in Bonita Creek were reproducing successfully and both fry and adults were in excellent condition and exhibiting excellent growth a year and a half after introduction. We attribute condition and growth to reduced number of fish relative to food supply (e.g. macroinvertebrates; Table 3).

Only four rainbow trout were captured in Ellison Creek in June 1992. Three of the four were in the release area and one was about 2 km downstream. No young-of-year rainbow trout were collected. On July 7, 1993, after a winter of significantly above average precipitation and runoff, Bonita and Ellison creeks were sampled again. Thirteen brook trout averaging 254 mm TL were captured and released in Bonita Creek. All had been displaced downstream to a reach of stream where only 1 trout was taken in October 1991. Sampling in Ellison Creek resulted in only 2 adult rainbow (TL 300 mm+). Young-of-year were collected in both streams, however.

Aquatic macroinvertebrates

Prior to the fire, estimated mean aquatic macroinvertebrate densities ranged from 5000-9000/ m^2 in the three streams within the burn area (Rinne 1996). Samples immediately after the fire and prior to either slurry or flood flow events indicated no significant effect on populations. Sampling in the three creeks following a slurry flow event (July 10), however, revealed a 7-fold to 10-fold reduction in macroinvertebrate density. Further sampling in late July following several significant flood events revealed populations were effectively reduced to zero. Sampling after winter flow events (May 1991) indicated populations had partially recovered. Analyses of additional samples collected in autumn 1991 and 1992 indicated population densities had not stabilized and were fluctuating below pre-fire densities. Macroinvertebrate diversity (number of taxa) was reduced from 25% (Bonita) to over 70% (Dude and Ellison) from pre-fire levels.

The Divide and McKnight Fires, New Mexico

In the 1980s, Main Diamond Creek, Gila National Forest, contained the largest population (*ca.* 5,000 individuals) of the endangered Gila trout (Mello and Turner 1980). The stream had many log structures added (Rinne 1981b) to increase pool habitat and sustain this native wild trout population in a marginal, headwater stream (Regan 1966). Based on size structure, the population in Main Diamond Creek appeared to be stunted, presumably caused by artificial creation of an over abundance of pool habitat (Regan 1966, Rinne 1982). By comparison, McKnight Creek (Gila National Forest) supported a population of Gila trout introduced from Main Diamond Creek that ranked second in total numbers of fish only to Main Diamond (Mello and Turner 1980).

A major objective of the Gila Trout Recovery Plan was to duplicate all five natural populations of this endangered trout (U.S. Fish and Wildlife Service 1984). Such a strategy was designed to reduce the probability that natural disturbance events such as drought, flooding, or wildfire would destroy any of these populations. The Recovery Plan stated, "While there is little danger that a single catastrophic event (e.g., forest fire) would ever exterminate all populations of Gila trout simultaneously, such an occurrence could seriously reduce numbers of the species and threaten efforts to secure its future. On the other hand, a catastrophe could exterminate one or more of the native populations, ..."

In July 1989 a lightning strike ignited a wildfire on the ridge between Main and South Diamond Creeks. Before it was extinguished, the Divide Fire consumed over 4,000 ha of forest vegetation on the watersheds and in the riparian areas (Propst et al. 1992). Because of the concern for elevated runoff and possible toxic flows of ash, an immediate, combined effort by the Forest Service and New Mexico Game and Fish removed 566 Gila trout and placed them in the U. S. Fish and Wildlife Service's Mescalero National Fish Hatchery for subsequent restocking. Initial slurry flows were similar to those described following the Dude Fire. However, large amounts of ash were transported into the stream (Propst et al. 1992). Two weeks later several kilometers of Main Diamond were sampled. A single,live Gila trout was found. Sampling about 1.5 km in October 1989 after extensive flooding as a result of denudation of the watershed, revealed no individuals of this endangered species. Electrofishing of the entire 6 km of previously inhabited portions of Main Diamond Creek in May 1990 again revealed no fish. Similar to Main Diamond, sampling of South Diamond Creek and its headwaters yielded only one Gila trout.

During the 1980s, a series of 50-100 year flood events occurred in McKnight Creek. These events

drastically changed channel morphology. Based on original introductions and periodic sampling by personnel at New Mexico State University indicated these floods greatly impacted the Gila trout population (Medina and Martin 1988 and Fig. 2). Such changes could be attributed to a wet cycle, increased runoff from the watershed, or the combination of both. Although not specifically studied, we suggest that the McKnight Creek stream channel may yet be adjusting to the large 20,000+ ha wildfire on the headwaters of McKnight Creek roughly four decades earlier (Medina and Martin 1988).

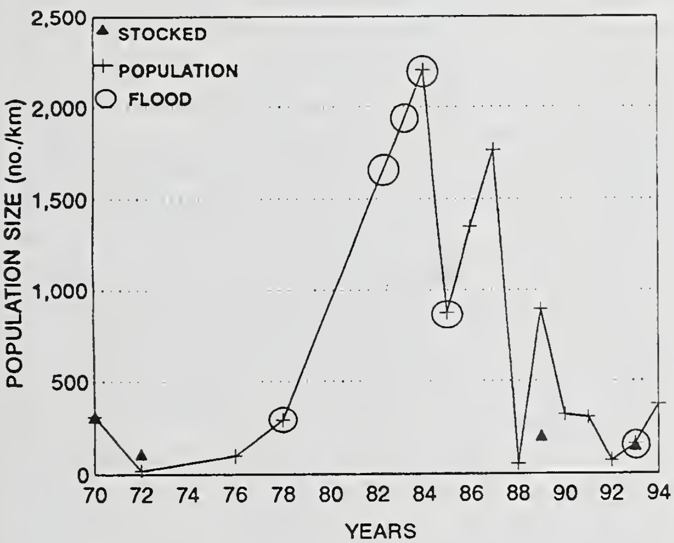

Figure 2. Fluctuations in Gila trout populations in McKnight Creek two to four decades after wildfire.

DISCUSSION

There is very little published information on the effects of either natural wildfire or prescribed fire on stream habitat and associated fisheries in the Southwest (Severson and Rinne 1990). Because of the marked effects of the aftermath of wildfires discussed above, and the amount of area burned annually by prescribed fire in the southwestern region, it is essential, for several reasons, that this lack of information be addressed. First, the demand for recreational fishing in the Southwest is high compared to national averages (Everest and Summers 1982). Equally important is the fact that the majority of the native fishes in the Southwest are threatened, endangered, sensitive, or of special concern and thereby are protected by law (Rinne and Medina 1995, Rinne and Minckley 1991). Federal agencies are mandated by the Endangered Species Act of 1973 to insure that Federal land management activities do not negatively impact or "take" federally listed species. Further, because Forest Service sensitive species are both numerous and are candidates for federal listing, they must also be wisely conserved. In combination, the economic demand for a recreational fishing resource and the legal mandate to protect a valuable, declining native fish fauna render it timely and essential to begin to address not only wildfire but prescribed fire effects on fishes as well.

Trout occupying marginal, headwater habitats such as those in Dude, Ellison, Bonita, Main Diamond and McKnight Creeks are particularly vulnerable to wildfire effects. Case histories suggest the probability is very high that total fish populations will be lost following a large wildfire. Although more subtle, cumulative, and unstudied the impacts following prescribed burns potentially affect both the short- and long-term habitats and food supply of fishes. Our data suggest that toxic slurry or ash flows are immediately fatal to a portion of the population. Survivors of these events become physiologically stressed. Flood flows following slurry flows then effectively extirpate surviving individuals. The extent of fish loss will depend both on burn intensity and size, frequency, and duration of flows during the summer monsoon. Springs or upwellings may serve as refugia, as apparently occurred for the one surviving brook trout recorded at the head of Dude Creek.

The marked effect of the aftermath of fire on salmonid species in these three case histories presents a paradox. "Why do any wild populations of native trout, or any other native species of fishes, occur throughout headwater streams within forested landscapes in the Southwest?" Fires have been occurring for centuries in these landscapes (Dieterich 1980, Krammes 1990). This paradox needs closer examination.

The population of Gila trout in Main Diamond Creek was the largest of all natural populations of this endangered species albeit possibly attributable to extensive stream habitat improvement structures. Nevertheless, this and four other natural populations (Mello and Turner 1980) have persisted through historic time. Based on the aftermaths of the Dude and Divide fires, one should not expect any native salmonids to be present in headwater streams in the Southwest. The native Apache trout was formerly

(late 1800s) so abundant in the White Mountains of Arizona that it was not unusual to catch 100 in an afternoon (Rinne 1985). Other native non-salmonid native fish species also persist in montane streams on National Forests in the Southwest (Rinne and Medina 1995). Considering historic (pre 1890) fire cycles (4-5 years) in the Madrean Province of the Southwest and northern Mexico (Dieterich 1980, Swetnam 1990) one would not expect any of these species to be present today in highly vulnerable, headwater habitats. It becomes apparent that "not all things equal" on our forested watersheds between a century or more ago and the present. The question needs to be asked "Why, in spite of historical fire patterns and studies that suggest total extirpation of populations following large wildfires, do any native fishes persist in southwestern riparian-stream ecosystems?"

Because forest landscapes historically consisted of more open stands of pine, fires were typically more frequent, less intense, and smaller. High intensity crown fires were a rare occurrence (Covington and Moore 1992). Wildfires were modified first by heavy grazing and then by fire suppression efforts. Increased fire suppression resulting in fuel loading on forest watersheds has resulted in the opposite effect—greater frequency of larger fires that are more intense, localized, and consume a greater percentage of watershed vegetation, and denude a greater percentage of landscape surfaces. Generally, fires burn in mosaics and consume only a portion of the total watershed vegetation. The Dude Fire was estimated to have burned an average of 65% of pre-fire vegetation; in certain areas on the Bonita and Ellison Creek watersheds it totally removed all vegetation.

Historically, the lower intensity, naturally caused wildfires were followed by the summer monsoon season, just as occurred on the Divide and Dude fires. However, with less fuel buildup and lower intensity, non-crowning fires, extensive toxic ash flows and massive flood event were less likely. The combination of less post-fire ash entering streams, and reduced likelihood of subsequent massive floods has permitted trouts and other native fishes to survive in montane, headwater habitats through time.

In a cumulative perspective, the current extensive use of prescribed fire as a forest management tool in the southwestern USA could be an important factor in influencing sustainability of fish populations and distributions. A prescribed burn (The Shannon Burn) near the Dude Fire (Medina and Baker 1996) and observed accumulations of sediment after a single flood event upstream of a newly-constructed weir on Dude Creek in summer 1991, suggest the potential cumulative effects of sediment on fishes and their food supply and habitats following even prescribed fire are potentially as great a threat as from wildfires.

Although not immediately fatal to salmonids in marginal headwater habitats, the amounts of sediment (< 2 mm) mobilized from watersheds following even small, low-intensity "control" burn such as the Shannon Burn could temporarily alter habitat for both fishes and their food supply. Cumulatively in time and space, this impact could become significant. Fine sediment effectively fills the interstices of substrate and ultimately reduces macroinvertebrate density (Bjornn, et al. 1977; Rinne and Medina 1988; Everest et al. 1987). This negative impact on food supply, combined with aggradation in pool habitats essential for fish survival during both drought and winter periods, may indeed be as great as after natural wildfire.

The fire on the McKnight Creek watershed occurred almost 4 decades ago. Further, Gila trout were not introduced into Mcknight until 1972, almost 20 years post-fire. The response of the Gila trout in Mcknight and those trout introduced following the Dude fire suggests that re-establishing salmonid populations in such marginal, headwater habitats will be a time consuming perhaps impossible management exercise. Any time frame will be a function of burn intensity, watershed size, flow intensity, and species of fish. Stream improvement structures were heavily impacted by flooding in the 1980s in McKnight Creek. The Gila trout population, although reduced and subject to marked fluctuations, continues to survive. Nevertheless, several decades may be required for watersheds subjected to severe wildfire to completely stabilize to the point of dampening extreme flow events and fluctuations in riparian-stream habitats and fish populations.

CONCLUSIONS

Fire has been a natural disturbance factor in Madrean forested landscapes for centuries. Prior to European settlement of the area, frequent, low intensity fires were common. These fires affected the watersheds that fed the streams containing both native salmonids, Apache and Gila trout, and other

native fishes. Occasionally these natural disturbance events severely impacted fish habitats and populations, but, not to the extent we are witnessing today.

With ever-increasing urbanization of forested montane areas in the Southwest (Medina 1990), a "prescribed natural fire" management philosophy to reduce fuel loads would probably lead to some hot, crowning wildfires that not only are potentially destructive to forests, private property, and human life, but to stream habitats and fishes.

Current fire management philosophy was severely tested during the 1988 Yellowstone Fires (Romme and Despain 1989; Philpot 1990; Varley 1990). Continuing to suppress wildfires in order to prevent destruction of private property or threat to human life needs re-examination relative to the natural, historical, wildfire frequency occurrences and vegetational succession. Permitting fuel build up through fire suppression only increases the probability of devastating wildfire. Romme and Despain (1989) present an excellent ecological/fire management scenario for the Yellowstone ecosystem. Except for tree species and successional cycles, the same scenario would be applicable to fire management in Madrean ecosystems.

Containment/confinement strategy of addressing wildfires is a basic component of fire management philosophy. On one hand, fuel buildups occur as fires are suppressed. On the other, prescribed fire is a management tool widely employed to reduce fuel loading and reduce the threat of holocausts such as the Dude, Divide, and Mcknight fires. These two fire management philosophies must be corroborated relative to each other and with management of two high priority management activities in the Southwest: riparian-stream habitats and fishes.

Select "ecological assessment" panels following the Yellowstone fires concluded that some kind of "natural-fire program" is appropriate and necessary for maintaining the wilderness value of parks and refuges. Although National Forests are typically managed for multiple uses rather than wilderness values, fire management of forested landscapes in the Southwest that is synchronized with natural processes (e.g. vegetational succession and fire ecology) has a greater probability of minimizing impacts on all renewable natural resources, including fishes.

Forest Service management and research must continue to cooperate in examining the potential effects of both suppression and prescription fire management strategies on riparian habitats and associated native and recreational fishery resources in the southwestern region. Such a partnership must be approached in the context of "naturalness" (Philpot 1990), "ecology-based resource management" of the U. S. Forest Service, and sustainability of ecosystems (Lubchenco, et al. 1991).

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Effects of Fire on Birds in Madrean Forests and Woodlands

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Abstract.—Fire usually affects birds indirectly, by altering habitat or food resources. Bird response may be positive or negative, depending on life-history characteristics and fire extent, intensity, and duration. Disruption of natural fire regimes may have far-reaching consequences for these birds and their habitat. The effects of fire on forest birds should be studied experimentally, at both species and community levels, in conjunction with efforts to restore fire as a natural process in Madrean forests. The effects of fire on important habitat components (e.g., snags, logs, and oaks) should be monitored, and the effects of salvage logging on post-fire bird communities should be carefully evaluated.

INTRODUCTION

Concern about the effects of fires on western forests and woodlands in recent years (Lotan and Brown 1985, Lotan et al. 1985, Krammes 1990) reflects a growing awareness that fire is a natural process in these ecosystems, and that suppressing fire has unintended consequences. In forest types normally subject to frequent low-intensity fires, fire suppression, coupled with heavy grazing, has resulted in increased stand density and fuel loading (Harrington and Sackett 1990; Covington and Moore 1994a,b; Sackett et al. 1994, 1996; Arno et al. 1995; Minnich et al. 1995; Touchan et al. 1995). Consequently, when fires do break out in these forests, they are becoming larger, more intense, and more likely to result in stand replacement than historical fires (Harrington and Sackett 1990, Swetnam 1990, Covington and Moore 1994a, Sackett et al. 1994).

The forests and woodlands of the Madrean Sky Island Archipelago of the southwestern United States and northern Mexico (Warshall 1995) are also subject to this trend. As a result of decades of fire suppression and grazing, intervals between fires, stand density, and fuel loadings have increased in these mountain ranges from encinal woodlands upward through pine-oak (*Pinus* spp. -*Quercus* spp.), pine, and mixed-conifer forests (Baisan and Swetnam 1990; Fulé and Covington 1994, 1995, 1996; Barton 1995; Caprio and Zwolinski 1995; Grissino-Mayer et al. 1995; Villanueva-Díaz and McPherson 1995).

The threat of stand-replacing wildfire is exacerbated in dry years (Swetnam 1990, Swetnam and Bettancourt 1990). For example, 1994 was an unusually severe fire year in the mountains of southeastern Arizona and southwestern New Mexico, with 589 fires burning more than 34,000 ha (84,000 acres; Allen 1995). The 1996 fire season promises to be even worse, with record low fuel-moisture conditions as a result of lack of winter precipitation.

Because the forests in the Sky Island Archipelago are restricted to isolated mountains, some of which are relatively small, a single large wildfire could damage or completely remove a considerable portion of the existing forest in some of these mountains. The danger of such wildfires is increasing yearly.

The Sky Island mountain ranges are important centers of biodiversity, due to the intermingling of northern and southern floral and faunal elements (Barton 1995, Felger and Wilson 1995, Warshall 1995). Organisms residing in Madrean forests and woodlands have evolved with fire as a natural process. The types of stand-replacing fires likely to occur under current conditions could have far-reaching effects on these organisms, however, becaue they are not similar to the fires that these organisms evolved with.

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Here, we summarize current knowledge on the possible effects of fires on the rich and unique avifauna (Marshall 1957, Felger and Wilson 1995, Warshall 1995) of the Madrean Sky Island Archipelago. Our treatment is limited to montane forests and woodlands, with the lower bound defined by encinal woodland (e.g., Warshall 1995: fig. 2). We first evaluate current knowledge on the effects of fire on forest birds in general, then review what is known about fire and birds in the Madrean Archipelago. Finally, we discuss some implications of this information for forest managers.

EFFECTS OF FIRE ON FOREST BIRDS IN GENERAL

Fire can affect forest birds positively or negatively, depending on the type and extent of fire and the particular life history of the species involved. Direct effects, such as mortality due to fire, are generally considered to be minor. Rather, fires influence birds indirectly through habitat modification, changes in food supply, or changes in abundance of competitors and/or predators (Rotenberry et al. 1995). The effects of fire on habitat structure, floristic composition, and food resources may be especially important, singly or in combination, as many birds respond strongly to these features of their habitat (MacArthur et al. 1966, Koplin 1969, Rotenberry 1985).

Several authors have reviewed the effects of fire on forest birds (Bendell 1974, Hutto et al. 1992, Dobkin 1994, Hejl 1994, Hejl et al. 1995, Rotenberry et al. 1995). These reviews suggest that generalizing about fire effects is difficult, for many reasons.

First, fires vary widely in extent, intensity, and duration (Rotenberry et al. 1995). Because of this variation in fire behavior, the effects of fire on birds and their habitat also varies widely.

Second, fire effects also vary across temporal scales. For example, cavity-nesting birds may respond positively to fire in the short term, but long-term effects may be negative as the burned snags fall (Raphael and Morrison 1987, Raphael et al. 1987, Hejl et al. 1995, Johnson and Wauer in press). Conversely, intense burns that greatly alter bird habitat in the short-term may be necessary for long-term maintenance of natural patterns of forest succession in some forest types (Hutto 1995).

Third, differences in the life histories of various bird species can result in different responses to fires. For example, cavity-nesting birds, timber-drilling birds, and granivores often respond positively to burns because of increased nesting substrates and/or food supplies (Lowe et al. 1978, Overturf 1979, Wauer and Johnson 1984, Hejl 1994, Hejl et al. 1995, Hutto 1995, Johnson and Wauer in press). In contrast, foliage gleaners may respond negatively (Roppe and Hein 1978, Overturf 1979, Blake 1982) due to decreased foraging substrate. Response patterns may even vary within guilds (Root 1967) in some cases (Skinner 1989, Hutto 1995, see also Mannan et al. 1984).

Finally, methodological problems plague many of the studies of the effects of fire on birds. Most studies have been conducted opportunistically rather than planned, limiting the inferences that can be drawn (Dobkin 1994:14, Hutto 1995). Most were also restricted in both spatial and temporal scale, and lacked the replication necessary to show general patterns (Dobkin 1994:14, Hutto 1995). Most studies have focused on breeding bird communities, and ignored wintering and migrating birds (but see Blake 1982). Many studies that reported few differences in bird communities between burned and unburned areas relied on composite statistics such as total bird abundance or species richness, rather than examining responses of individual species. Because individual species may respond in opposite fashion, such composite measures may hide rather than reveal patterns (Mannan et al. 1984., Rotenberry 1985, Hejl et al. 1995, Hutto 1995). Finally, even studies that have evaluated responses of individual species typically have not examined demographic parameters (Hejl 1994). Burned areas could contain many birds, yet function as population sinks if reproduction is insufficient to balance mortality (Robinson 1992).

The point of the above discussion is not to denigrate past studies, but simply to note the many difficulties involved in documenting general patterns with respect to the effects of fires on forest birds. Nevertheless, some broad generalizations are possible; note that all of these are somewhat dependent on the spatial scale of the observations.

First, patterns differ with burn intensity, being most pronounced for intense burns. Cavity-nesting birds, timber-drilling birds, granivores, and some aerial insectivores often respond positively to intense burns in the short term (Hejl et al. 1995, Hutto 1995) due to increases in perching, feeding, and nesting substrates. A few species, such as the Black-

backed Woodpecker (*Picoides arcticus*; species names for birds follow AOU 1983, 1995) are nearly restricted to intense burns, and may require such burns for long-term population maintenance (Hutto 1995).

Effects of low- and moderate-intensity burns are less dramatic. In the short-term, bird species richness may increase in moderate-intensity burns, because birds characteristic of both burned and unburned forest may use the area (Taylor and Barmore 1980). Low-intensity burns can create or maintain habitat for species that prefer open forest (Marshall 1963, Hutto 1995). Thus, some species are favored by low-intensity burns, whereas others are favored by high-intensity burns. In general, fire suppression has probably resulted in declines of birds that use snags preferentially in burned areas (Hejl 1994; see also Brawn and Balda 1988), and may have reducednumbers of some open-forest species as well (Marshall 1963).

EFFECTS OF FIRE ON BIRDS IN MADREAN FORESTS

Few studies are available documenting responses of birds to fire in the Madrean Archipelago. Marshall (1963) noted that fire regime, habitat conditions, and bird communities all varied in parallel between the mountains of southern Arizona and northern Mexico (Sonora and Chihuahua). Fires had been effectively suppressed in Arizona, but not in Mexico. As a result, forests and woodlands in Arizona were denser than similar types in Mexico. Several bird species common to brush or dense forest, including the Elf Owl (*Micrathene whitneyi*), Ash-throated Flycatcher (*Myiarchus cinerascens*), Blue-gray Gnatcatcher (*Polioptila caerula*), Black-throated Gray Warbler (*Dendroica nigrescens*), Scott's Oriole (*Icterus parisorum*), and Spotted Towhee (*Pipilo maculatus*), were more abundant in Arizona than in Mexico. In contrast, several species preferring open forest conditions were more abundant and/or occurred at higher elevations in Mexico, presumably because open forest conditions persisted to a higher elevation there. These included the American Kestrel (*Falco sparverius*), Cassin's Kingbird (*Tyrannus vociferans*), Curve-billed Thrasher (*Toxostoma curvirostre*), Common Nighthawk (*Chordeiles minor*), Purple Martin (*Progne subis*), Chipping Sparrow (*Spizella passerina*), and both Eastern (*Sialia sialis*) and Western (*S. mexicana*) Bluebirds.

Bock and Bock (1990:54-55) summarized the effects of two "cool" fires on birds and/or vegetation in open oak savannah (encinal) on the Appleton-Whittell Research Ranch, in the foothills of the Huachuca Mountains. This area is at the lower elevational bound discussed here. Fires burned in February and May. Little mortality of oak trees was observed, and shrubs were reduced in height but not in density. Grass cover was reduced in both burns, but recovered on one area within two years. Seed production increased in the first year following the fire. On one burned area, bird numbers were about 18% higher following the fire than on adjacent unburned areas (Bock et al. 1976). Most of this difference was attributable to increased abundance of two seed-eating birds, Mourning Doves (*Zenaida macroura*) and Chipping Sparrows. The Grasshopper Sparrow (*Ammodramus savannarum*), which depends on heavy grass cover, disappeared entirely from the burned area for the duration of the study. Bock and Bock (1990:55) concluded that much more research was needed on the effects of fire in lowland encinal on birds and their habitat.

Horton and Mannan (1988) studied the effects of prescribed burning on snags and cavity-nesting birds in pine-oak forest in the Santa Catalina Mountains, Arizona. They sampled abundance of snags in different size and decay classes and bird abundance before and after burning in three stands. They also sampled three control (unburned) stands.

The prescribed burn resulted in a moderately-intense surface fire which remained within prescribed limits. Nearly half of all ponderosa pine (*P. ponderosa*) snags ≥15 cm in diameter at breast height were burned down or drastically altered. Because few large trees were killed immediately, there was a net 45% decrease in large snags in the first season following the prescribed burn.

Few differences were observed in bird populations before and after fire. Only Northern Flickers (*Colaptes auratus*) and Violet-green Swallows (*Tachycineta thalassina*) declined in abundance in burned stands, and only Mountain Chickadees (*Parus gambeli*) increased. Horton and Mannan (1988) concluded that observed declines in cavity-nesting birds (Northern Flicker and Violet-green Swallow) were not due to a shortage of nest sites, because post-fire snag densities exceeded densities theoretically required to support pre-fire populations of cavity-nesting birds.

MANAGEMENT IMPLICATIONS

Restoring Fire to the System

Fire is obviously important as a natural process in Madrean forests and woodlands, and there are compelling ecological reasons to restore fire to these systems (Baisan and Swetnam 1990; Harrington and Sackett 1990; Covington and Moore 1994a; Fulé and Covington 1994, 1995, 1996; Sackett et al. 1994, 1996; Grissino-Mayer et al. 1995). Recent experience suggests that if low-intensity fire is not restored to these forests, sooner or later they will be subject to stand-replacing wildfire (Covington and Moore 1994a, Sackett et al. 1994). Such wildfires may not be totally negative as far as birds are concerned. They may be beneficial in areas where they burn in a mosaic pattern, or for some species that require intense burns. Stand-replacing wildfires can also destroy habitat for many species, however. For example, Johnson and Wauer (in press) observed "diverse, substantial populations of breeding birds" even in areas subjected to severe crown fire. They also noted that recovery of bird populations following fire was delayed in areas of high-intensity fire, however.

The substantial economic costs involved in fighting intense wildfires provide another incentive for restoring more natural fire regimes. For example, consider two fires on the Gila National Forest, New Mexico (data on file, Gila National Forest Supervisor's Office, Silver City, NM). The Pigeon Fire (June/July 1994) burned approximately 3,240 ha (8,000 acres) in the Aldo Leopold Wilderness, despite active suppression efforts conducted at a cost of roughly $7,500,000. In contrast, the Bonner Fire (June/July 1995), 8 km (5 mi) north of the Pigeon Fire, was allowed to burn under the Prescribed Natural Fire (Mutch 1995) program. This fire burned a much larger area (approximately 11,540 ha [28,500 acres]), but the cost for monitoring this fire (approximately $100,000.) was far lower than the cost to fight the Pigeon Fire. Further, the expensive efforts to fight the Pigeon fire were unsuccessful; both fires were extinguished by rain. Under more natural fire regimes, we anticipate that more fires could be allowed to burn naturally, substantially reducing the economic costs of fire suppression (Fulé and Covington 1996, Sackett et al. 1996).

Restoring more natural conditions in these forests and woodlands is problematic given current conditions. Heavy fuel loadings that have accumulated as a result of fire suppression can result in hot burns that kill overstory trees. Harrington and Sackett (1990) reported 35% mortality in old-growth ponderosa pines following prescribed burning in northern Arizona. These trees had survived numerous presettlement burns, but were unable to survive the first burns in 100 years due to the excessive fuels that had accumulated. This suggests that several applications of cool fire may be required in many areas to reduce fuels without killing overstory trees. Relatively frequent maintenance burns will also be required to prevent new buildup of fuels (Harrington and Sackett 1990; Sackett et al. 1994, 1996). These repeated cool burns may result in more smoke and particulate matter than the public is currently willing to accept (Daniel 1990, Lahm et al. 1990), and may result in conflicts with existing air quality standards (Chambers and Duncan 1985). Thus, restoring more natural conditions in these forests will take not only time and money, but also considerable education of the public on fire as a process and the costs (both economic and ecological) of fire suppression.

Evaluating the Effects of Fire on Forest Birds

Having argued for restoration of natural fire regimes in the Sky Islands, we must also point out that it may not be possible to accomplish this without some impacts to their unique avifauna. To minimize such impacts, we must know more about the effects of fire on birds in Madrean forests and woodlands. Recent (and future) intense wildfires will provide opportunities for monitoring bird community composition through various stages of post-fire succession. This may be the only feasible approach to studying effects of large stand-replacing fires on birds. For low- to moderate-intensity fires, however, we believe that more could be learned through an experimental approach using prescribed burning.

Researchers should work with land managers to design and conduct studies documenting the effects of a range of fire prescriptions on birds and their habitat. In evaluating the effects of fire on birds, both species- and community-level responses should be considered (Hejl et al. 1995, Hutto 1995), along with demographic parameters (Hejl 1994, Dobkin 1994) and patterns of resource use. Particular efforts should be made to identify any species that may be depen-

dent on or sensitive to fire. Important habitat components that should be evaluated include snags, logs, oaks, and under- and mid-story vegetation. Snags are important as feeding and nesting sites, particularly for woodpeckers and other cavity-nesting birds (Koplin 1969, Balda 1975, Scott 1979, Cunningham et al. 1980, Raphael and White 1984, Horton and Mannan 1988, Hejl et al. 1995, Hutto 1995). Down logs can also serve as feeding sites (Horton and Mannan 1988), and several species of oaks provide important resources for forest birds in the Madrean Archipelago (Block et al. 1992). Ground-nesting birds and species that primarily forage in the under- and mid-story may be particularly sensitive to the effects of fire on under- and mid-story vegetation.

Some or all of these habitat components may be vulnerable to fire (e.g. Horton and Mannan 1988, Bock and Bock 1990, Barton 1995) and special protective measures may be required to maintain them at adequate levels. Both managers and the public must realize and accept that there may be short-term declines in some of these habitat components despite such measures, however.

Ideally, studies of fire effects could be designed in conjunction with efforts to restore fire as a natural process in these forest types. Such efforts should aim to reduce fuel loads in the short term, with a long term goal of returning these systems toward natural disturbance regimes. This would require mimicking natural disturbance patterns in terms of frequency, intensity, and extent. Unless variation in timing, intensity, and scale of fires is incorporated in prescribed burns, such burns are unlikely to closely mimic natural fire patterns (e.g., DesGranges and Rondeau 1993).

Evaluating the Effects of Salvage Logging on Forest Birds

The effects of salvage logging on post-fire bird communities should also be carefully evaluated. Snags can provide important resources for birds in general (Balda 1975, Scott 1979, Cunningham et al. 1980, Taylor and Barmore 1980, Dickson et al. 1983, Raphael and White 1984, Brawn and Balda 1988, Raphael et al. 1988) and specifically for post-fire bird communities (Koplin 1969, Taylor and Barmore 1980, Raphael and White 1984, Horton and Mannan 1988, Hejl et al. 1995, Hutto 1995). For example, woodpeckers may concentrate in burned areas to feed on insects in snags (Koplin 1969, Wauer and Johnson 1984, Hutto 1995, Johnson and Wauer in press) and may be recruited to such areas over long distances (Wauer and Johnson 1984). Thus, retention of snags could mitigate the effects of wildfires on forest bird communities (Moeur and Guthrie 1984, Hutto 1995, Johnson and Wauer in press). Retaining significant groups of snags would be preferable to retaining scattered isolated snags, both to minimize short-term snag loss to windthrow, and to provide concentrated food sources for woodpeckers. Preference should also be given to retaining large snags that existed prior to the burn, as these are used more often for feeding and nesting than case-hardened snags resulting from the burn (Moeur and Guthrie 1984, Hutto 1995).

CONCLUSIONS

Clearly, we know far less about the effects of fire on birds than we would like. Further, most of what we do know relates to areas or forest types outside of the Madrean Sky Islands, and may not be directly applicable to the Sky Island avifauna. This lack of knowledge is particularly troubling because:

1. The avifauna of the Sky Islands is unique;
2. Fire is important as a natural process in the Sky Islands; and
3. Some of the forests in this area are relatively small and thus highly vulnerable to impacts from intense wildfires.

We know too little about the effects of fire on the avifauna of the Sky Islands to accurately predict the effects of particular types of fire on that avifauna. Because these species evolved with fire as an important process shaping their habitats, we believe that restoration of natural fire regimes is both desirable in these areas and compatible with maintaining this unique avifauna. The process of restoring more natural conditions may result in declines in some habitat components and/or some avian species. We suspect that these impacts will be short-term, and outweighed by the long-term benefits. Impacts to forest birds and their habitat should be minimized where possible, however, by approaching restoration cautiously, by carefully monitoring bird populations and habitat conditions, and by paying special attention to any avian species that appear to be declining as a result of

burning. In particular, efforts should be made to identify thresholds of acceptable decline, and to ensure that no avian species decline beyond the point from which it would be difficult for them to recover.

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Faunistic Diversity in Four Mountain Ranges in Northeast Sonora, México: Field Notes

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Abstract.—Field notes and comments are presented about faunal diversity in four mountain ranges in the Madrean Archipelago in Northeast Sonora, México, observed during a formal survey on Mexican Spotted Owl from January, 1993 to December, 1995. Of 253 wildlife species reported for the area, only 146 were recorded: 7 amphibians, 21 reptiles, 81 birds, and 37 mammals. Some important observations include the black bear, a recent record of a new black tailed prairie dog colony, Mexican spotted owl sightings, and some Mexican Goverment Listed species.

INTRODUCTION

Global faunistic diversity includes 43,853 chordate species, of which 4,184 are amphibians, 6,300 are reptiles, 9,040 birds, and 4,000 mammals, for a total of 23,524 wildlife species, excepting fishes (Wilson, 1988). Many of these species may now be extinct or under a threatened or endangered status.

In comparision, Sonora State in México, with several biomes represented, has recorded 189 amphibian and reptile species, almost 500 species of birds and 120 mammals, according to the Conservation Data Base (Centro de Datos para la Conservación or CDC) from Centro Ecológico de Sonora, 1993; Mc Craine and Wilson, 1987; Monson and Phillips, 1981; Leopold, 1977; Marshall, 1957; Moore, 1938). Species listed as threatened or endangered by the Mexican goverment include: Sonoran pronghorn (*Antilocapra americana sonoriensis*), porcupine (*Erethizon dorsatum*), black bear (*Ursus americanus*), black-tailed prairie dog (*Cynomys ludovicianus arizonensis*), jaguar (*Panthera onca*), ocelot (*Felis pardalis*), yagoarundi (*Felis yagoarundi*), otter (*Lutra canadensis*), masked bobwhite quail (*Colinus virginianus*), thick billed parrot (*Rynchopsita pachyrhyncha*), military macaw (*Ara militaris*), golden eagle (*Aquila chrysaetos*), bald eagle (*Haaliaetus leucocephalus*), peregrine falcon (*Falco peregrinus*), aplomado falcon (*Falco femoralis*), Mexican spotted owl (*Strix occidentalis lucida*), clapper rail (*Rallus longirostris*), desert tortoise (*Gopherus agassizii*), Gila monster (*Heloderma suspectum*), ridge-nose rattlesnake (*Crotalus willardii*), "chicotera" snake (*Masticophis flagellum*), Chiricahua frog (*Rana chiricahuensis*), and other species (SEDESOL, 1994. The Mexican gray wolf (*Canis lupus baileyi*), grizzly bear (*Ursus horribilis*), imperial woodpecker (*Campephilus imperialis*), beaver (*Castor canadensis*), and others are considered to be extinct (Hoffmeister, 1986).

In the madrean montane archipelago, in Northeast Sonora, Marshall (1957) reports 70 bird species for Sierra Los Ajos Mountain range, 47 in Sierra de Cananea, and 67 in the Sierra San Luis area; CDC (1993) reports 253 wildlife species, with 15 amphibians species, 51 reptiles, 118 birds, and 69 mammals. According to SEDESOL (1994), 56 of these species are under a protection category, with 4 amphibians, 22 reptiles, 16 birds and 14 mammals.

There are not enough recent studies on effects of fire on wildlife species in the study area.

STUDY AREA

The area is located in Northeast Sonora, north of the Sierra Madre Occidental. Isolated mountains surrounded by valleys dominate the area. There are many such mountains, but the study was restricted to four main mountain ranges: Sierra de Cananea, Sierra Los Ajos, Sierra La Púrica, and Sierra San Luis (fig. 1). The biggest mountain range is Sierra Los

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Figure 1. Map of Sonora State.

Ajos, which jointly with Sierra Buenos Aires and Sierra La Púrica forms the "Sierra Los Ajos, Buenos Aires y La Púrica National Forest Reserve" with a total area of 21,494 ha.

Elevation ranges from 1,100 meters above sea level (msl) in the valleys and small hills to 2,640 msl at Cerro de Las Flores (Sonora's highest peak) in Sierra Los Ajos; 2,520 msl at S. Mariquita and 2,580 msl in S. Elenita, 2,460 msl at La Púrica Mountain and 2,500 msl at S. San Luis.

Weather is very similar in the four mountain ranges. The year-round average temperature is about 15.9°C and the average rainfall is 523.8 mm in the Cananea area, with maximum rainfall during August and September (Garza, 1993).

Topography varies from plains and small hills in the valleys to deep canyons with rough and steep cliffs in the mountains.

Vegetation in valleys and small hills includes several grass species, *Yucca* sp, alligator juniper (*Juniperus deppeana*), oaks (*Quercus* spp), palmilla (*Nolina* spp.) and other plants. There are some mesquite intrusions (*Prosopis juliflora*). Hillsides and middle elevations support several associations, with oaks (*Quercus emoryi, Q. viminea, Q. arizonica, Q. oblongifolia*), mexican pinyon (*Pinus cembroides*), madrone (*Arbutus arizonica*) and manzanita (*Arctostaphyllus pungens*). In the higher areas, it is common to find ponderosa pine (*Pinus ponderosa*), ayacahuite pine (*P. ayacahuite*), Apache pine (*P. engelmannii*), Arizona pine (*P. arizonica*), chihuahua pine (*P. leiophylla*), velvet ash (*Fraxinus velutina*), Gambel oak (*Qercus gambellii*), silverleaf oak (*Q. hypoleucoides*), Douglas-fir (*Pseudotsuga menziessii*), and other species. Also, in the high slopes with north exposure, there are white fir (*Abies concolor*) and quaking aspen (*Populus tremuloides*)(Fishbein, Felger y Garza, 1994; Solís, Brady and Medina, 1993; Solís, 1987; Garza, 1985; Lehr, 1978; Elmore and Janish, 1976; White, 1943).

Sierra de Cananea (Elenita and Mariquita) is a ridge of three hydrologic basins in Sonora: San Pedro flows to the Gila River; Sonora River and Cocóspera River flow to the Magdalena-Concepción river system. Sierra Los Ajos divides the water flow to the Bavispe River to the east and Sonora River to the west. Sierra La Púrica discharges water to the Bavispe and Moctezuma rivers, both tributaries to the Yaqui River; Sierra San Luis also discharges to the Bavispe River and to the plains in Chihuahua State. All the ephemeral, intermittent, and permanent creeks of these ranges support riparian forest composed of: Arizona sycamore (*Platanus wrightii*), sugar leaf maple (*Acer grandidentatum*), velvet ash (*Fraxinus velutina*), Arizona cypress (*Cupressus arizonica*), Arizona walnut (*Juglans major*), Arizona alder (*Alnus oblongifolia*), alligator juniper (*Juniperus deppeana*), Rocky Mountain juniper (*J. scopulorum*), some oak species(*Quercus* spp), and other species (Garza, 1993; Lehr, 1978).

METHODS

The present study was derived from field notes taken from January 1993 to December 1995, during the development of a Mexican spotted owl survey and during the planning and design of three natural protected areas in Northeast Sonora in the program: "Sistema de Areas Naturales Protegidas del Estado de Sonora" (SANPES): Sierra Mariquita-Elenita-Río San Pedro, Sierra Los Ajos, Buenos Aires y La Púrica, y Sierra San Luis. A total of 27 field trips, averaging 12 days, were made during 1993, 1994, and 1995.

Species were observed directly along field transects by sightings, songs, and calls, or indirectly by pellets, nests, burrows, tracks, skins, bones, or carcasses.

Ten stainless steel Sherman traps and four home-made wire traps were placed randomly in the field on four of the 27 fieldtrips, to confirm small mammals and rodents in the area. Peanut butter and banana were used as bait. For bird capture, 15 mist nets were used during three field trips; oat, corn and barley were used as bait. For reptiles, an herpetological hook and a small canvas sack were used. On two occasions, a fish spoon net was used to capture bats. Any captured animal was managed with care to avoid damage or stress, and liberated later in the same place where taken.

Skulls, bones, and carcasses were collected directly from the field, and occasionally were received from ranchers and local people. An extensive bibliography was reviewed to determine fauna historical records for comparison with the fieldtrip notes.

RESULTS

Of 253 wildlife species reported in northeast Sonora, 57.7% were recorded in the present study: 146 species in four mountain ranges (appendix and table 1).

At Sierra Los Ajos we detected a total of 120 wildlife species, while in Sierra de Cananea, Sierra La Púrica, and Sierra San Luis, the numbers of identified wildlife species were 93, 92, and 90, respectively (table 2).

SPECIES OF INTEREST DETECTED IN THE STUDY AREA.

Black bear (*Ursus americanus*)—Presence recorded in four sightings on four mountain ranges. Observations of mortality included two kills in the Sierra La Purica and three in the Sierra San Luis. Bear move off higher ranges after fire to lower elevations, where they are often killed by ranchers.

Black Tailed Prairie Dog (*Cynomis ludovicianus arizonensis*)—A prairie dog population of almost 200 individuals, distributed in 5 colonies, was found in the San Pedro River Valley, near the Arizona-Sonora border. Hofmeister (1986) considered this subspecies to be extinct in Sonora since 1938.

Mexican gray wolf (*Canis lupus baileyi*)—Some unconfirmed sightings and hearings by local people in the Cananea area.

Porcupine (*Erethizon dorsatum*)—None were detected in the area in the present study, but some are said to be in the Sierra Los Ajos.

Ringtail (*Bassariscus astutus*)—Four individuals were detected, two in Sierra Los Ajos, one in Sierra La Púrica, and one near Esqueda, northeast of S. La Púrica.

Golden eagle (*Aquila chrysaetos*)—An adult was detected in the north part of Sierra la Mariquita.

Peregrine falcon (*Falco peregrinus*)—A pair were detected in southeast Sierra Los Ajos, and a single individual in Sierra La Púrica.

Aplomado falcon (*Falco femoralis*)—A male was detected in the east foothills of Sierra Mariquita, while a pair were detected in the San Pedro River Valley and a solitairy individual was recorded in the lower part of Arroyo Los Ajos in the mountain range with same name.

Common black hawk (*Buteogallus anthracinus*)—Two pairs were detected, one in El Apache Canyon at Sierra Los Ajos and the other North of Sierra La Púrica.

Mexican spotted owl (*Strix occidentalis lucida*)—Several individuals were detected, as pairs or singles,

Table 1. Species observed in the study area, by taxonomic group, compared with numbers reported in the literature(CDC,1993)

Taxonomic Group	Recorded species in the study area	Reported species by CDC (1993)
Amphibians	7	15
Reptiles	21	51
Birds	81	118
Mammals	37	69
TOTAL	146	253

Table 2. Recorded species in four mountain ranges. Numbers in parentheses indicate number of species in various protection categories.

Taxonomic group	Cananea	Los Ajos	La Púrica	San Luis
Amphibians	5 (1)	6 (2)	1 (0)	3 (0)
Reptiles	7 (2)	13 (9)	13 (8)	7 (3)
Birds	63 (9)	74 (10)	57 (7)	54 (8)
Mammals	18 (5)	27 (7)	21 (4)	26 (6)
TOTAL	93 (17)	120 (28)	92 (19)	90 (17)

in all the mountain ranges, except Sierra San Luis, which has historical records.

Ridge-nose rattlesnake (*Crotalus willardi obscurus*)—This species was detected in Sierra San Luis and Sierra Los Ajos.

Tiger salamander (*Ambystoma tigrinum*)—Some individuals were recorded in the middle part of El Quince Canyon at Sierra Mariquita, and in El Huérigo Creek, Los Tecolotes, and El Alazán Canyons at Sierra Los Ajos.

SPECIES DETECTED IN BURNED AREAS.

Several wildlife species were detected in areas affected by old and recent fires:

Puerto El Charro, Sierra Los Ajos—Old fire area: *Odocoileus virginianus, Meleagris gallopavo, Strix occidentalis, Micrathene whitneyi, Chordeiles minor*, and small rodents.

Cordón de Arriba, Cañón de Evans, Sierra Los Ajos—Area with fire in 1982; *Strix occidentalis, Cyrtonix montezumae, Cathartes aura, Columba livia, Melospiza melodia, Ursus americanus, Felis concolor, odocoileus virginianus*, and others.

El Manzano, Cañada Hoya del Packard, Sierra Los Ajos—Near El Manzano at Joya del Packard Canyon, in an area with fire in 1990:*Ursus americanus, Tayassu tajacu, Odocoileus virginianus, Melagris gallopavo, Buteo jamaicensis, Aphelocoma ultramarina, Cyanocitta stelleri, Carpodacus mexicanus*, and others.

Los Jacalitos, Southeast Cabral Mountain, Sierra Los Ajos—1986 fire area: *Coragyps atratus, Meleagris gallopavo, Melanerpes formicivorus, Odocoileus virginianus* and *Nasua narica*.

Center-East Peak, North Slope, Sierra La Púrica—*Odocoileus virginianus, Felis concolor, Bassariscus astutus, Meleagris gallopavo, Picoides scalaris, Melanerpes formicivorus, Sceloporus clarki*, and others.

Higher area, Cañada El Quince, Sierra Elenita—*Ursus americanus* (female with a cub), *Corvus corax, cathartes aura, Cardinalis cardinalis*, other species.

Center-West Canyon (no name), Astronomic Observatory, Sierra Mariquita—*Strix occidentalis, Parabuteo unicinctus, Odocoileus virginianus, Corvus corax*, and others.

Las Chimeneas Canyon, San Antonio Ranch, Sierra San Luis—*Ursus americanus, Felis concolor, Tayassu tajacu, Urocyon cinereoargenteus, Nasua narica, Melanerpes formicivorus, Pyrocephalus rubinus, Sialia sialis, Coragyps atratus*, and others.

Cañada del Diablo, West slope, Sierra San Luis—This is the fire area with fewest species detected, just one ridge-nose rattlesnake (*Crotalus willardi obscurus*) and a group of seven coatimundi herd (*Nasua narica*).

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APPENDIX

List of wildlife recorded in northeast Sonora (based on the 1995 CES Conservation Data Base).

Codes: P, risk of extinction; A, threatened; R, rare; PR, special protection

AMPHIBIANS

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
AMBYSTOMATIDAE					
Ambystoma tigrinum	PR	X	X		
BUFONIDAE					
Bufo alvarius		X			
Bufo cognatus			X		
Bufo debilis	R				
Bufo punctatus		X	X		X
Bufo woodhousii					
HYLIDAE					
Hyla arenicolor					
LEPTODACTYLIDAE					
Eleutherodactylus augusti					
MICROHYLIDAE					
Gastrophryne olivacea	R				
PELOBATIDAE					
Scaphiopus bombifrons					
Scaphiopus couchii		X	X	X	X
Scaphiopus multiplicatus					
RANIDAE					
Rana catesbeiana					
Rana chiricahuensis	A				
Rana tarahumarae			X		X

BIRDS

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
ACCIPITRIDAE					
Aquila chrysaetos	P	X			X
Buteo albonotatus					
Buteo jamaicensis	PR	X	X	X	X
Buteo regalis					
Buteogallus anthracinus	A		X	X	
Circus cyaneus	A	X	X		
Parabuteo unicinctus	A	X	X	X	X
AEGITHALIDAE					
Psaltriparus minimus		X	X	X	X
APODIDAE					
Aeronautes saxatalis		X	X		
BOMBYCILLIDAE					
Bombycilla cedrorum		X	X	X	

BIRDS—CONT'D

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
CAPRIMULGIDAE					
Chordeiles acutipennis					X
Chordeiles minor			X	X	
Phalaenoptilus nuttallii		X	X	X	X
CATHARTIDAE					
Cathartes aura		X	X	X	X
Coragyps atratus		X	X	X	X
CERTHIIDAE					
Certhia americana					
COLUMBIDAE					
Columba livia		X	X	X	
Columbina inca			X		
Zenaida asiatica		X	X	X	X
Zenaida macroura		X	X	X	X
CORVIDAE					
Aphelocoma ultramarina		X	X	X	X
Corvus corax		X	X	X	X
Corvus cryptoleucus		X	X	X	X
Cyanocitta stelleri		X	X	X	X
CUCULIDAE					
Geococcyx californianus		X	X	X	X
EMBERIZIDAE					
Agelaius phoeniceus		X	X	X	X
Aimophila botterii					
Aimophila cassinii			X		
Aimophila ruficeps		X	X	X	X
Ammodramus bairdii					
Ammodramus savannarum					
Cardellina rubrifrons					
Cardinalis cardinalis		X	X	X	X
Cardinalis sinuatus		X	X	X	X
Dendroica graciae					
Dendroica nigrescens					
Dendroica petechia			X		
Geothlypis trichas					
Icterus galbula		X	X		
Junco phaeonotus					X
Melospiza lincolnii					
Melospiza melodia		X	X		
Molothrus aeneus		X	X	X	X
Molothrus ater		X	X	X	X
Myioborus pictus	R	X	X		X
Oporornis tolmiei					
Peucedramus taeniatus					
Pipilo chlorurus					

BIRDS—CONT'D.

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
EMBERIZIDAE—Cont'd.					
Pipilo erythrophthalmus					
Pipilo fuscus		X	X	X	
Quiscalus mexicanus		X	X	X	X
Sturnella neglecta					
Vermivora celata					
Vermivora virginiae					
Xanthocephalus xanthocephalus			X	X	X
Zonotrichia leucophrys		X	X	X	X
FALCONIDAE					
Falco columbarius	A	X			
Falco femoralis	A	X	X		
Falco mexicanus					
Falco peregrinus	A		X	X	
Falco sparverius		X	X	X	X
FRINGILLIDAE					
Carpodacus cassinii		X			
Carpodacus mexicanus		X	X	X	X
Loxia curvirostra		X	X	X	X
HIRUNDINIDAE					
Hirundo rustica		X	X	X	X
Tachycineta thalassina		X	X		X
LANIIDAE					
Lanius ludovicianus		X		X	X
MIMIDAE					
Mimus polyglottos		X	X	X	X
Toxostoma curvirostre		X	X	X	X
MUSCICAPIDAE					
Polioptila melanura		X	X	X	
Sialia currucoides					
Sialia mexicana		X	X		X
Turdus migratorius			X	X	X
PARIDAE					
Parus wollweberi		X	X	X	X
PASSERIDAE					
Passer domesticus		X	X	X	X
PHASIANIDAE					
Callipepla gambelii		X	X	X	X
Callipepla squamata					
Cyrtonyx montezumae		X	X	X	
Meleagris gallopavo		X	X	X	X
PICIDAE					
Colaptes auratus		X	X		X
Melanerpes formicivorus		X	X	X	X
Melanerpes uropygialis			X	X	X
Picoides scalaris		X	X		X
Picoides stricklandi					X
Picoides villosus					
PTILOGONATIDAE					
Phainopepla nitens		X	X	X	X
SITTIDAE					
Sitta carolinensis			X		
Sitta pygmaea					
STRIGIDAE					
Asio flammeus	A				
Asio otus					
Athene cunicularia	A		X		X
Bubo virginianus	A		X	X	X
Glaucidium gnoma	R				
Micrathene whitneyi	P	X	X		X
Otus flammeolus					
Otus trichopsis					
Strix occidentalis	A	X	X	X	
TROCHILIDAE					
Calypte anna				X	
Cynanthus latirostris					
Eugenes fulgens		X	X		X
Lampornis clemenciae					
Selasphorus platycercus					

BIRDS—CONT'D.

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
TROGLODYTIDAE					
Campylorhynchus brunneicapillus				X	X
Catherpes mexicanus		X	X	X	X
Thryomanes bewickii			X	X	X
Troglodytes aedon		X	X	X	X
TROGONIDAE					
Trogon elegans		X	X		X
TYRANNIDAE					
Empidonax difficilis					
Myiarchus cinerascens		X	X	X	
Myiarchus tyrannulus					
Pyrocephalus rubinus		X	X	X	X
Sayornis nigricans			X	X	
Tyrannus crassirostris					
Tyrannus verticalis					
Tyrannus vociferans		X	X	X	X
TYTONIDAE					
Tyto alba		X	X		X
VIREONIDAE					
Vireo bellii	P				
Vireo solitarius					

MAMMALS

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
CANIDAE					
Canis latrans		X	X	X	X
Canis lupus	P				
Urocyon cinereoargenteus		X	X	X	X
CERVIDAE					
Odocoileus hemionus	A		X		X
Odocoileus virginianus		X	X	X	X
DIDELPHIDAE					
Didelphis virginiana			X	X	
ERETHIZONTIDAE					
Erethizon dorsatum	P				
FELIDAE					
Felis concolor		X	X	X	X
Lynx rufus		X	X	X	X
GEOMYIDAE					
Thomomys bottae			X		X
Thomomys umbrinus			X		
HETEROMYIDAE					
Dipodomys merriami	A	X	X		X
Dipodomys ordii				X	X
Dipodomys spectabilis					
Perognathus baileyi			X	X	
Perognathus flavus					X
Perognathus hispidus					
Perognathus intermedius					
Perognathus penicillatus					
LEPORIDAE					
Lepus alleni	R				
Lepus californicus	R	X	X	X	X
Sylvilagus audubonii		X	X		X
Sylvilagus floridanus		X	X	X	X
MOLOSSIDAE					
Eumops perotis					
Tadarida brasiliensis			X		
MORMOOPIDAE					
Mormoops megalophylla					
MURIDAE					
Baiomys taylori					
Microtus mexicanus					
Neotoma albigula	A	X	X		X

MAMMALS—CONT'D.

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
Neotoma mexicana				X	
Onychomys leucogaster					
Onychomys torridus					
Peromyscus boylii					
Peromyscus eremicus				X	
Peromyscus leucopus			X		X
Reithrodontomys fulvescens					
Reithrodontomys megalotis					
Reithrodontomys montanus					X
Sigmodon hispidus		X			
Sigmodon ochrognathus					X
MUSTELIDAE					
Conepatus mesoleucus				X	X
Mephitis macroura		X			
Mephitis mephitis		X	X	X	X
Mustela frenata					
Spilogale putorius					
Taxidea taxus	A	X	X	X	X
PHYLLOSTOMIDAE					
Choeronycteris mexicana	A				
Leptonycteris sanborni	A				
Macrotus californicus					
PROCYONIDAE					
Bassariscus astutus	A		X	X	
Nasua nasua		X	X	X	X
Procyon lotor			X	X	X
SCIURIDAE					
Sciurus arizonensis	A				
Spermophilus spilosoma		X	X	X	X
Spermophilus variegatus					X
SORICIDAE					
Notiosorex crawfordi	A		X	X	
TAYASSUIDAE					
Tayassu tajacu		X	X	X	X
URSIDAE					
Ursus americanus	P	X	X	X	X
VESPERTILIONIDAE					
Antrozous pallidus					
Eptesicus fuscus					
Lasiurus borealis					
Lasiurus cinereus					
Myotis californicus					
Myotis thysanodes					
Myotis velifer		X			
Myotis volans					
Myotis yumanensis					
Pipistrellus hesperus					
Plecotus townsendii					

REPTILES

FAMILIY/Species	Status	S.Can.	S.Ajos	S. Púrica	San Luis
COLUBRIDAE					
Diadophis punctatus					
Gyalopion canum					
Gyalopion quadrangulare	R				
Heterodon nasicus	R				
Hypsiglena torquata	R		X	X	
Lampropeltis getulus	A				
Lampropeltis pyromelana	A		X	X	X
Masticophis bilineatus		X	X		
Masticophis flagellum	A			X	
Pituophis melanoleucus					
Rhinocheilus lecontei					
Salvadora grahamiae					
Salvadora hexalepis					X
Sonora semiannulata					
Tantilla wilcoxi					
Tantilla yaquia		X			
Thamnophis cyrtopsis	A		X		
Thamnophis eques	A				
Thamnophis marcianus	A				
CROTALIDAE					
Crotalus atrox	PR	X	X	X	
Crotalus lepidus	PR		X		
Crotalus molossus	PR		X	X	
Crotalus priceii	PR				
Crotalus scutulatus	PR	X	X	X	X
Crotalus willardii	PR		X		X
EMYDIDAE					
Terrapene ornata	PR		X	X	
Trachemys scripta	PR				
HELODERMATIDAE					
Heloderma suspectum	A				
IGUANIDAE					
Cophosaurus texanus	A				
Crotaphytus collaris	A		X		
Gambelia wislizeni	R				
Holbrookia maculata					
Phrynosoma douglassi					
Phrynosoma modestum					
Phrynosoma solare				X	
Sceloporus clarki		X	X	X	
Sceloporus jarrovi			X	X	X
Sceloporus magister					
Sceloporus scalaris					
Sceloporus virgatus					
Urosaurus ornatus					
Uta stansburiana		X	X	X	X
KINOSTERNIDAE					
Kinosternon flavescens					
Kinosternon sonoriense		X	X	X	X
LEPTOTYPHLOPIDAE					
Leptotyphlops dulcis					
SCINCIDAE					
Eumeces obsoletus					
TEIIDAE					
Cnemidophorus burti					
Cnemidophorus exsanguis					
Cnemidophorus tigris		X			
Cnemidophorus uniparens					
TESTUDINIDAE					
Gopherus agassizii	A			X	

Desert Bighorn Sheep and Fire, Santa Catalina Mountains, Arizona

Paul R. Krausman, George Long, and Luis Tarango[1]

Abstract.—We studied the influence fire had on visibility for desert bighorn sheep (*Ovis canadensis mexicana*) in Pusch Ridge Wilderness, Arizona. We mapped fires that occurred from 1956 to 1987 and randomly selected burned and unburned sites for visibility measurements. Overtime visibility decreased when the areas were not burned. Increased vegetation in areas that were not burned may be detrimental to desert bighorn sheep habitat. Land managers should allow wildfires to burn in and adjacent to desert bighorn sheep habitat if fire enhances visibility for the species.

INTRODUCTION

The role of natural wildland fires in desert ecosystems and its effects on desert bighorn sheep (*Ovis canadensis mexicana*) is not well documented (Krausman et al. 1979). Geist (1971) and Peek et al. (1979) indicate that bighorn sheep habitat is characterized by long lasting climax grassland communities. Bighorn sheep use subclimax grasslands that are generated by fires and show strong preference for habitats providing good visibility (Risenhoover and Bailey 1980).

Pusch Ridge Wilderness (PRW) in the Santa Catalina Mountains contains an indigenous herd of desert bighorn sheep. Reductions in visibility within the habitat have been attributed to contributing to a restricted range of habitat (Etchberger et al. 1989, Gionfriddo and Krausman 1986).

Gionfriddo and Krausman (1986) described this habitat as an open tree invaded semidesert grassland. This vegetation type is described by Lowe (1964) as the Encinal and Mexican Oak (*Quercus* spp.)-Pine (*Pinus* spp.) woodland and Sonoran desertscrub (upland subdivision). Very little literature and research has been published dealing specifically on the effects of burning within these vegetation associations. More data are needed to better understand the role fire plays to shape these plant communities and the animals that depend upon them.

Krausman et al. (1979) and Etchberger et al. (1989) recount some of the fire history of the Santa Catalina Mountains. They describe what role fire might play to explain the vegetation composition, thermal cover, and visibility characteristics of occupied and abandoned bighorn sheep ranges in the PRW. Etchberger et al. (1990) studied a single fire event and the effect it had upon composition and thermal cover in the currently occupied mountain sheep range in the PRW. Their research indicated that after 6 months, post-fire vegetation conditions were not significantly different than pre-fire conditions with the exception of thermal cover that was reduced. Etchberger et al. (1989) believed that their vegetation data indicated a fire induced seral vegetation gradient. However, details about a historical fire frequency that produce this gradient are not known or is it quantified.

None of these researchers discusses the short or long term effects a historical fire frequency has had on vegetation associations within this bighorn sheep range and how the distribution of the PRW sheep population might change as new open grassland areas are created or closed by successional changes. Geist (1971) postulates a fluctuating habitat theory that states mountain sheep will usually stay within a confined area of traditional use unless new habitat is made available adjacent to the existing habitat. Peek et al. (1985) states that "...areas of prime bighorn habitat where wildfire was common were incorporated into designated wilderness areas, and a knowl-

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edge of bighorn response to fire was needed if fire was to be restored to the ecosystem." More detailed information of the historical fire frequency for this area is needed to better understand bighorn sheep responses to fire induced vegetation changes.

Recent research indicates bighorn sheep use of burned ranges is related to 2 factors: the increased amounts of forage protein made available early in spring by greening vegetation following fire versus the slower 'greenup' time of unburned ranges (Bentz and Woodard 1988), and bighorn sheep use of burned areas that provide horizontal views unobstructed by dense vegetation and reduced amounts of overstory cover (Bentz and Woodard 1988).

Researchers believe bighorn sheep rely more on sight distance than their other senses to avoid predators and will, therefore, forage and use more efficiently those areas that afford the greatest security from predators (Berger 1978, Wakelyn 1987). Wakelyn (1987) studied the abandonment of Rocky Mountain bighorn sheep (*Ovis canadensis canadensis*) ranges and found abandoned bighorn sheep habitat is in part attributed to increased amounts of shrubland resulting in decreased horizontal visibility. This was attributed to fire suppression policies that increased forest (Wishart 1978) and shrub cover to the point of decreasing interpreted visibility values by 54 percent in 15 years (Wakelyn 1987). Although Wakelyn (1987) states that the loss of mountain sheep habitat due to fire suppression activities for numerous years has not been fully assessed; the aforementioned discoveries indicate that bighorn sheep populations may be regulated in part by the amount of suitable habitat that affords good sight distances created by the occurrence of natural wildfire. Our objective is to evaluate the effects that fire has had on the vegetation communities and bighorn sheep visibilty in the PRW.

STUDY AREA

The study area was within historical bighorn sheep range found in the Santa Catalina Mountains. The PRW lands are administrated by the U. S. Forest Service, Coronado National Forest, Santa Catalina Ranger District and are located adjacent to Tucson, Arizona. This area was estimated to have approximately 50 to 100 desert bighorn sheep (Etchberger et al. 1989). This herd was unique because it was an indigenous population. According to Etchberger et. al. (1989) this population occupies only 44 km2. Mountain sheep habitat in PRW has been described by Etchberger et. al. (1989) and Gionfriddo and Krausman (1986) as steep rugged terrain with a large range in topographic relief. Etchberger et al. (1989) also found the occupied range to have long distances to human disturbances, good visibility, sideoats grama (*Bouteloua curtipendula*), red brome (*Bromus rubens*), brittle brush (*Encilia farinosa*), high amounts of forbs and low amounts of shrubs.

METHODS

The areas we analyzed were in proximity to adequate escape terrain within the elevation range suitable to the current mountain sheep population. We also characterized the recent fire history of the Santa Catalina Mountains. This information was then used to identify specific fire incident locations for comparison to known unburned areas of similar slope, aspect, and elevation covering the same time period.

Using available Forest Service records (U. S. Forest Service, Fire Atlas of the Santa Catalina Ranger District, Arizona, unpubl., 1989) we delineated accurate fire locations for fires that occurred from 1921 to 1989. We used aerial photos (U.S. Forest Service, aerial photo files, Coronado Natl. For., Ariz., 1954-1987) to confirm the location of fires mapped in the U. S. Forest Service's Santa Catalina Ranger District Fire Atlas. We mapped all Forest Service fires > 4 ha. When available, pre- and post-fire aerial photo vegetation textural and reflectance changes were used to confirm Fire Atlas mapping of fire polygons. Fire locations were then mapped on 7.5 minute United States Geological Survey topographic guadrangles (Fig. 1).

Known single fire occurrence incidence covering 3 fire periods (1956, 1975, 1987) were selected and sampled based upon their location within the currently used mountain sheep range.

We selected the burned and unburned sample areas based upon similar slope, aspect, elevation and proximity within an arbitrary limit of 1.6 km for comparable unburned sample areas (Fig. 2). All unburned vegetation transects were then grouped for comparison to individual burn areas.

Habitat components such as rock, bare ground, and litter cover were recorded along 50 m linear

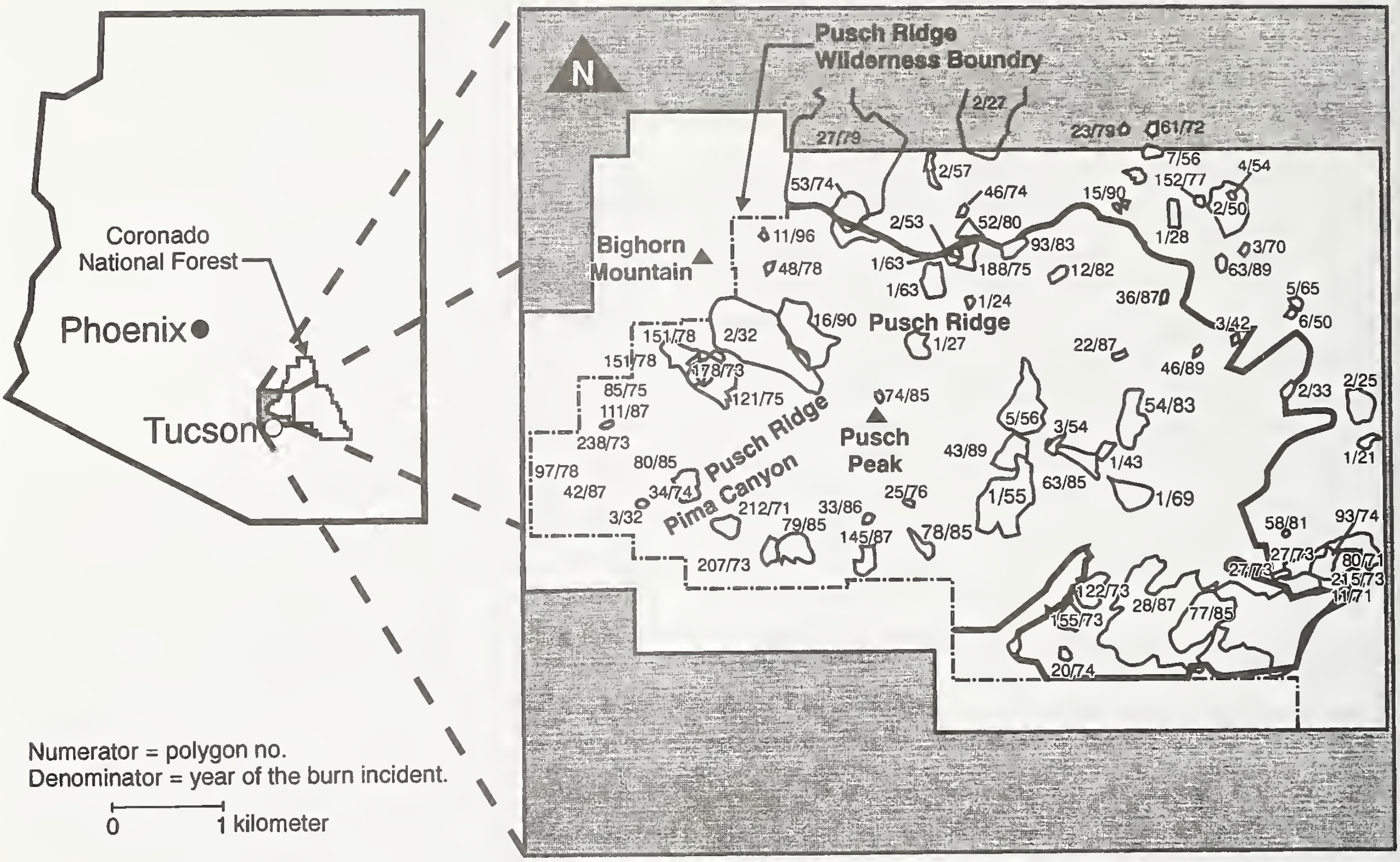

Figure 1. Compiled burns ≥ 4.04 ha in the Santa Catalina Mountains and Pusch Ridge Wilderness, Arizona, 1921-1989.

transects. Elevation, slope, aspect, distance to escape cover, and visibility at 20 and 40 m distances were measured at the beginning and end of each transect (Risenhoover and Bailey, 1980). Slope was measured using a clinometer and measured to the nearest whole percent. Visibility measurements included ocular estimates of the percentage of visibility blocked by undulations in topography and dense vegetation. All measurements were rounded to the nearest whole number with the exception of elevation data.

Fire frequency data were obtained using unpublished fire history data collected by the U. S. Forest Service. Only 15 years of computerized fire history data were available for frequency analysis on the Coronado National Forest. These data were retrieved using the "Qwick Qwery" (version 6.1) program accessible from the National Fire History Library database located in Fort Collins, Colorado. It is a computerized compilation of all reported fire incidence of both naturally ignited and man caused fire from 1970 to 1989.

Fire frequency analysis in this study only analyzes naturally ignited fire (lightning) data forest-wide for 1970 to 1985. Forest Service fire records estimate fire origins only in even (152 or 304 m) increments. The slope measurements used to compare areas were limited in interpretation to even increments of 0 to 9 percent starting at 0 percent and ending with 100+ percent. Vegetation classifications found in the available fire records used a general dominance classification, (i.e., grassland, ponderosa pine, [*Pinus ponderosa*]) and were extrapolated for their probability of occurring within or adjacent to desert bighorn sheep range. These forest-wide data were then extrapolated to imply the potential for fire frequency within bighorn sheep range on PRW. The fire frequency data were then compared to the results of the vegetation structure of past fires specifically sampled in this study. No effort was made to isolate only those fires that occurred in PRW.

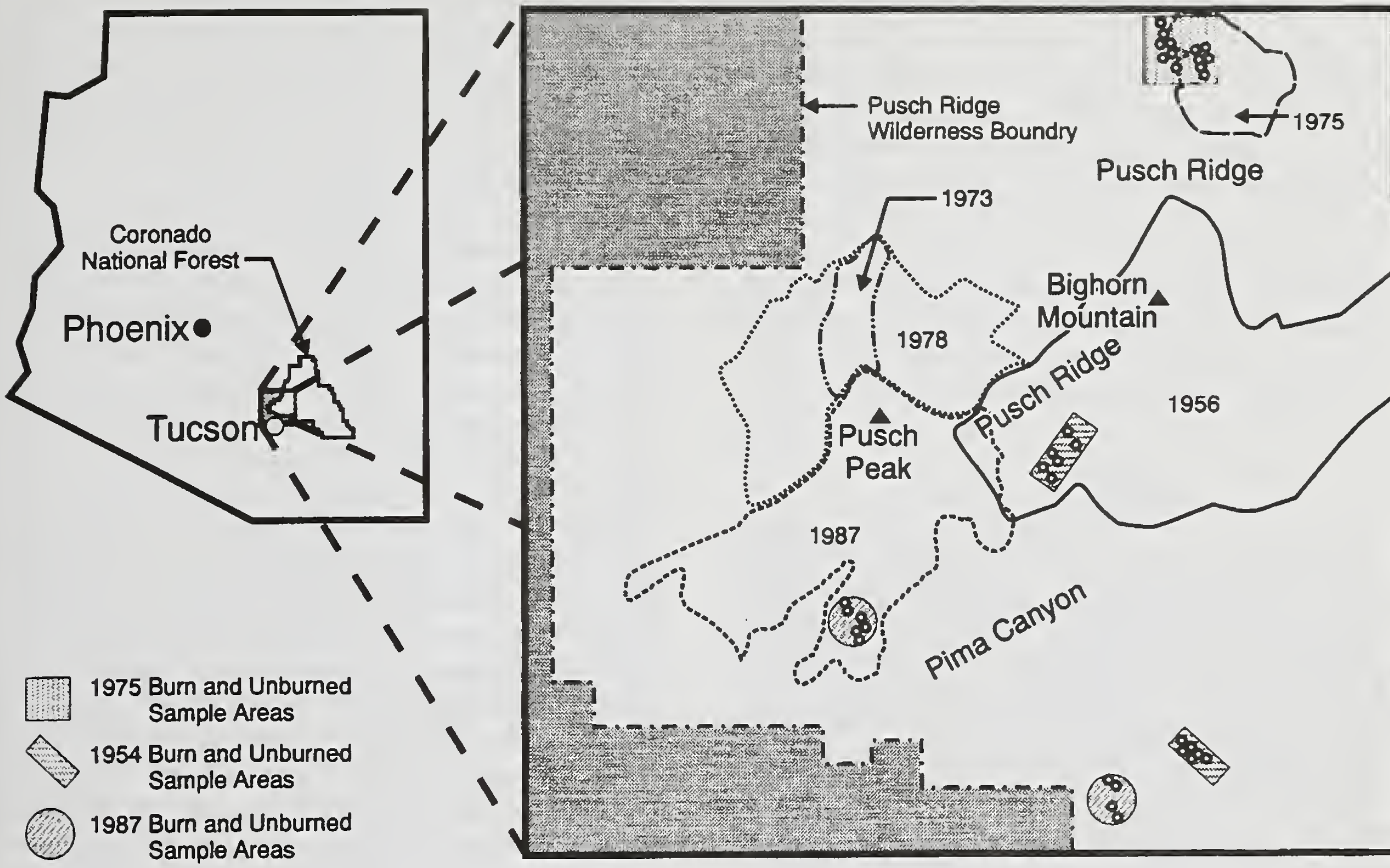

Figure 2. Locations of areas that were burned to contrast with unburned areas for visibility measures, Pusch Ridge Wilderness, Arizona.

RESULTS

All burned and unburned sites had comparable aspects (220°–246°) and elevations (1,194–1,240 m). Transect sites in the 1956 burn averaged 10 percent less slope than all the other sites. All areas sampled contained adequate escape cover within 150 m as specified for bighorn sheep summer habitat by Gionfriddo and Krausman (1986).

Visibility (Table 1) was 15-22 percent higher at 20 m than 40 m for all sample areas. Visibility decreased 8-9 percent per burning period or 0.5 percent/year over 31 years. The highest visibility (49 percent) was found in the 1987 burn samples and the lowest (24 percent) was found in the unburned areas (Table 1).

Vegetation obstructions to visibility were always 1-3 percent lower at 20 m that at 40 m. The highest amount of vegetation obstruction was found on the

Table 1. Visibility and obstruction means for desert bighorn sheep measured at 20 and 40 m "in the Pusch Ridge Wilderness, Santa Catalina Mountains, Arizona."

Year of burn	No. transects	Percent visibility		Percent obstruction from slope		Percent obstruction from vegetation	
		20 m	40 m	20 m	40 m	20 m	40 m
1956	6	46	31	35	47	19	22
1975	5	52	37	35	47	13	16
1987	6	71	49	18	39	12	13
Unburned plots	16	34	24	32	43	34	35

unburned sample areas that were 22 percent higher for 20 and 40 m view distances when compared to the remnant vegetation obstructions found in the 1987 burn area. Even the 1956 burn area had 13-15 percent lower vegetation obstruction than the unburned area.

Slope obstructions ranged from 39 to 47 percent for the 40 m distances and 18-35 percent for the 20 m distances, or it is 25 to 50 percent the difference between the two view distances for all sample areas. The percent slope obstructions stayed constant for each of the viewing distances measured in both burned and unburned sample areas.

DISCUSSION

There was a steady decline in visibility from 1987 to 1956 in known burn periods and into areas that are known not to have been burned prior to 1956. Visibility measurements were highest in most recent burned areas and lowest in the unburned areas. Vegetation obstructions averaged over the 32-year period do increase approximately 0.20-0.33 percent a year and vegetation obstructions appear to be inversely proportional to visibility with a steady trend of increasing vegetation obstruction from 1956 to 1987. Vegetation obstructions almost doubled over the 32 year period studied and confirms the suspicions of Krausman et al. (1979), Wakelyn (1987) and Etchberger et al. (1990) that vegetation visual obstructions can increase the amount of abandoned bighorn sheep habitat. This means that areas once burned with an association converting fire, if not maintained with periodic fires, will become unsuitable to bighorn sheep in approximately 30 years depending what the lower limit of habitat visibility of bighorn sheep will tolerate.

Slope obstructions stayed constant and did not change over time, which is what would be expected. Land forms usually do not change unless there is some cataclysmic or severe erosion event. Rather this measure reflects uniformity of the local topographic characteristics shared by all sample sites and hints at a lower limit or obstructed visibility bighorn sheep would tolerate within their historic range for slope and vegetation. Forage areas with vegetation that obstructs visibility to the same level as slope effects would then be used to the same degree provided escaped terrain is within a comfortable distance.

Analysis of the fire frequency data for grassland, which would afford bighorn sheep the greatest visibility, showed that 7.6 percent of the fires resulted in a burned area > 2.02 ha in size by the time the fires were contained by the Forest Service. These fires could be considered to be adequately maintaining small portions of the habitat in a suitable condition for desert bighorn sheep use.

Aggregating all lightning fire occurrences for the brush, ponderosa pine, mixed conifer and oak woodland vegetation results in approximately 41.2 percent of all fires occurring in these associations that could presumably, without suppression activities, result in conversion to a subclimax grassland stage that would be more beneficial to desert bighorn sheep. This figure does not distinguish between fire sizes. This is because of their potential to become >2.02 ha in size had it not been for existing fire suppression policies that effectively limited many of them from becoming larger in this elevation zone.

The annual average number of fires that occur in each vegetation association for the elevation zone in which bighorn sheep might occupy of all lands within the Coronado National Forest are as follows: grasslands have approximately 24.6 fires, brush 10.8, woodlands 2.9, ponderosa pine 2.7, and mixed conifer 0.4 fires/year. According to the fire frequency literature overview provided by Wright and Bailey (1982), the commonly accepted large acreage vegetation converting fire frequency in each of these vegetation types is 30 years for desert grasslands, 20 years for brush (Arizona chaparral), 4.8 to 11.9 years for ponderosa pine, 5 to 12 years for mixed conifer, and 10 to 30 years for woodlands (pinyon-juniper).

A truer fire frequency for these vegetation associations would require detailed mapping of each fire in each vegetation association for the 15 year fire record to determine how often a fire 2.02 ha would occur at that elevation zone and vegetation type. The analysis provided is not conclusive evidence that desert bighorn sheep rely completely upon fire occurrences to maintain suitable habitat. However, the data does indicate that without fire suppression policies there is a 41.2 percent potential to reduce overstory vegetation associations back to a subclimax seral grassland stage that bighorn sheep prefer.

MANAGEMENT IMPLICATIONS

Wildlife habitat managers should strive for prescribed or natural wildfire that achieve complete

vegetation association conversion to be of benefit for desert bighorn sheep populations. Such fires will require a hot prescription and are difficult to control. If feasible, native seed sources should be collected and secured from the burn area prior to the burn because hot fires generally destroy soil stored seed of both desired and undesirable species. These seed sources could then be used to reseed and rejuvenate the post fire area at a much faster rate than would occur naturally from adjacent unburned seed sources. Rogers and Steele (1980) indicate that native seed sources may take from 5-20 years to recover to preburn conditions by its own regenerative sources.

Large amounts of shrub seeds stored in the soil should be expected in overgrown bighorn sheep ranges. Prescribed fires would optimally be conducted after two good precipitation years with burning ideally occurring during the driest part of the third year. Rogers and Vint (1987) found fires usually burned larger areas after two consecutive years of above normal winter precipitation. This will ensure there is sufficient fuel loading to carry an intense fire and a dry soil subsurface that is least likely to protect undesirable shrub root crowns and seed sources. Periodic maintenance fires of prescribed burns should initially be conducted within 5 years of the first burn and then at intervals of at least every 10 years thereafter. Peek et al. (1985) states that bighorn sheep should suffer only minimal effects in areas of high fire frequency.

Wildlife managers should also consider competition from other ungulates when prescribing or allowing natural wildfire to occur in and adjacent bighorn sheep ranges. Steuter and Wright (1980) inferred that white tailed deer (*Odocoileus virginianus*) use was reduced in areas that appeared to have low horizontal vegetation density (i.e., good visibility). Brown and Henry (1981) also found that white-tailed deer fawn survival is decreased during droughts and climates characterized by unreliable rains. An implication of this is that conversion of brushy vegetation associations back to seral subclimax grasslands dries out the sites and creates a higher evaporation/ transpiration ratio than if the area was still covered in shrubs. Drying out he site thus reduces competition between desert bighorn sheep and white-tailed deer which occur in the PRW area.

Forage competition between bighorn sheep and peccary may also be reduced, if it exists, by use of fire. Everitt et al. (1984) found cacti species will make up the majority of peccary diets and when cacti is absent in an area they will switch to mesquite pods and forbs. Fire however, is very effective at removing cacti from an area. Cacti surviving in adjacent unburned areas may cause peccary to shift their forage use patterns and home range away from bighorn range in search of cacti.

Other researchers, Davis et al. (1984) and DeBano et al. (1984) found fire vegetative cover recovery took 10 to 11 years to return to approximately 90 percent prefire cover in Arizona Chaparral. DeBano et al. (1984) found resprouting shrubs took 3 years to regain 33 percent of prefire cover. Also in Arizona Chaparral, Pase and Ingebo (1965) reported that shrubs sprouted vigorously after fire occurrence and took 5 years to reach approximately 66 percent of the prefire cover conditions.

A possible explanation for the difference in results may be that the fire that was studied by Etchberger et al. (1990) was a cool fire and not an association converting fire. This would resulted in only a partial kill of visually obstructing vegetation. Residual killed vegetation may not have been consumed and remained standing to obscure visibility in post fire measurements.

Possible compounding this would be the increased productivity often associated with fires in the remaining live plants that would increase shoot heights above that which is normal in the immediate postfire years. Cave and Patten (1984) found biomass to greatly increase in surviving and newly established plants after fire occurrences. Merrill et al. (1980) found burned area productivity was twice that of unburned areas. These authors found extensive brittle brush seedling establishment 9 months after the fire. In addition Rogers and Steele (1980) found most resprouting was by woody shrubs and least by cacti.

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Fire Effects on Spittlebug Populations on Buffelgrass Pastures in the Sonoran Desert

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Abstract.—Prescribed burns during the summer removed dead material and litter and disrupted the spittlebug (*Aeneolamia albofasciata* Lall.) life cycle. Buffelgrass (*Cenchrus ciliaris* L.) total green biomass production was greater when fire was applied immediately after the initiation of the summer rainy season. Early summer burns when the insect is between the second and the third nymphal stages, reduced the nymph and adult populations by 95% and 100%, respectively. Buffelgrass densities increased when burns were applied before egg hatch, immediately after egg hatch and between de 2 and 3 nymphal instars. Plant densities decreased when burns were applied after buffelgrass began summer growth and when buffel was unburned.

INTRODUCTION

Where natural fires no longer occur, prescribed fire can be used to achieve multiple management objectives. On rangelands prescribed fire has been used to reduce fuel loads, prepare a seedbed, control plant diseases, suppress woody plants, enhance herbage yields, increase forage availability, improve forage quality, manage livestock and wildlife grazing (Weaver and Tomanek 1951, Weaver and Albertson 1956, Davis 1959, Hardley and Kieckhefer 1963, Komarek 1963, Marshall 1963, Anderson et al. 1970, Vogl 1971, Dodge 1972, Wright 1972, Wright and Bailey 1980, Ibarra et al. 1986, and White and Hanselka 1991).

Prescribed fire has been used to control insect pest (Schmid and Parker 1990). It has sucessfuly been used for spittlebug (*Aeneolamia albofasciata* Lalleman) populations in sugar cane (*Saccharum officinarum* L.), pangola grass (*Digitaria decumbens* Stnt), and stargrass [*Cynodon plestostachyus* (Kschum) Pielger] along the Gulf of Mexico (Coronado 1978). Ibarra and Enkerlin (1974), and Enkerlin and Morales (1979) reported that burning kills spittlebug nymphs and the eggs located at or near the soil surface.

Morphological characteristics generally determine a plant's response to fire. In general, perennial grasses are better adapted to survive fire than are forbs or woody plants. During dormancy grasses and forbs escape fire damage because meristematic tissue is below the soil surface. Seasonal burning can exploit differences in growth patterns between warm- and cool-season grasses and forbs by promoting one group over another (Johnson 1970, White 1980, White and Hanselka 1991). For example, cool-season grasses such as threeawn (*Aristida spp.*), are damage when burned during active spring growth. Warm-season grasses burned at the same time may not be damaged because they are dormant.

In northeastern Mexico, prescribed fire during spittlebug life cycle may limit spittlebug populations and enhance buffelgrass growth. The objectives of this study were to determine if summer burning during the spittlebug life cycle would reduce insect populations and stimulate forage production.

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STUDY SITE

The study site is located 82 km north of Hermosillo in northwestern Sonora, Mexico (29° 41' N lat., 115° W. long.) on the Carbo Livestock Research Station. Elevation is 470 m, slope is 1-2%, and soil is a Anthony fine loam (Thermic Typic Torrifluvent). Soils are recent alluvium, weathered from granitic rocks, moderately basic (pH = 8.5-8.9), abd range from 2 to 6 m in depth (Hendricks 1985).

Average annual precipitation is 320 mm (Centro de Investigaciones Pecuarias del Estado de Sonora 1989). Precipitation is bimodally distributed with approximately 60% occuring as rain between July and September and the remaining 40% as rain between October and April. May, June and September are usually dry months but exceptions do occur. Daytime temperatures average 34°C in summer, but frequently exceed 40°C in June and July. Nighttime temperatures average 8°C in winter, but drop to 0°C in January and February.

METHODS AND MATERIALS

A 10 ha stand of shrub-free dense buffelgrass, naturally infested with spittlebug, was fenced to exclude livestock in the summer of 1985. Twenty, 50-by-50 m plots were established in a randomized complete block design with four replication. The study was conducted in the summers of 1985 and 1986. The burning treatments corresponded to five different stages in the life cycle of the spittlebug: 7 to 14 days before summer rains in late June (T_1); during or after eggs hatch or after the the accumulation of 50 mm of summer precipitation (T_2); between the second and third nymphal instar (T_3); and between the fifth nymphal instar and adult stage (T_4). The fifth treatment was unburned control (T_5).

Firelines were dozed around each plot. A backfire was used to remove vegetation in a 3 to 5 m strip and the remaining was burned with a headfire. The time from ignition to total consumption of all above-ground fuel was recorded for each plot. Burns were conducted between 9 and 10 a.m. (Table 1). Wind speeds varied from 8-12 km/h, air temperatures 29-32°C, and relative humidity fluctuated from 38-70%.

Plants were sampled in five 1- by 1-m randomly selected quadrats prior to and after burning, at the peak of the summer growing season (middle of August) during four consecutive years. Buffelgrass plants were clipped at 5 cm above the soil surface and forage was separated into live (green), recent-dead standing (yellow) and old-dead standing (gray) biomass components. Samples were dried in a forced-draft oven at 40°C for 72 hr. All yields are reported on a dry weight basis.

Stand density in each plot was estimated in five randomly selected 1 m^2 quadrats, these were marked prior to and after burning to counted the number of buffelgrass plants/m^2.

Spittlebug populations were sampled before fire was applied (1984 and 1985) and in August during four consecutive summers. In each plot nymphs were counted in five previously unsampled random, 1 m^2 quadrats between mid-June and August. Spittlebug were separated in five nymphal stages (Bodegas

Table 1. Fuel characteristics and environmental conditions at burn time during 1985 and 1986 in northwestern Mexico.

Season	Fuel load Kg/ha	Fuel water content %	Wind speed km/hr	Air temperature °C	R.H %
Summer 1985					
Prehatching (T_1)	3587	25	8.0	29.5	46
Hatching (T_2)	3247	38	8.1	32.0	43
2nd & 3rd nymphal instar (T_3)58	3378	39	12.1	35.0	38
5th nmphal instar & adults (T_4)	3287	45	12.5	29.0	66
Summer 1986					
Prehatching (T_1)	4973	35	7.9	30.0	44
Hatching (T_2)	4509	38	8.0	31.2	48
2nd & 3rd nymphal instar (T_3)	4305	42	9.6	33.8	52
5th nymphal instar & adults (T_4)	4689	50	9.5	32.2	70

1973). Insects were netted in 25 strokes, whilewalking in a zig zag fashion from each plot. Adult insectsfrom the 50 nets sweeps were averaged for each plot and the mean was considered a plot replication. Plot were netted every other day while adult insects were present.

Precipitation and temperature (maximum and minimum) were recorded daily at the Carbo Livestock Research Stationabout 2 km from the study site.

The experimental design was a randomized complete block with four replications for each burn applied at strategic times within the spittlebug life cycle. The accumulation of herbage within a component was highly variable among years. For example, old-dead standing biomass did not accumulate for four years after burning. Under these conditions the population variances were tested for homogeneity. When populations had common variances, the data was pooled and subjected to analysis; when population variances differed ($P \leq 0.05$), unburned plots were analyzed separately from burned plots. When F-values were significant ($P \leq 0.05$) Least Significant Differences test (Steel and Torrie 1960) were used to separate means.

Adult and nymph densities were averaged over all collection days for each plot and they were subjected to log transformation (x-1). However, insect densities are presented untransformed. Analysis was a split-plot ANOVA. Data from the two years of treatment were pooled since there were no difference in the response buffelgrass and spittlebug population by the year of burning (1985 and 1986).

RESULTS AND DISCUSSION

All the fuel characteristics and environmental conditions at burn time during 1985 and 1986 are showed on table 1. The prescribed burn coincided with specific spittlebug development and buffelgrass phenological stages. The phenological stages of buffelgrass at the time of each burn were when (a) no growth, (b) at second leaf stage (5 cm), (c) early culm elongation (10-15 cm) and (d) active growth (leaves > 20 cm).

Burning prior to buffelgrass growth (T_1) removed old above-ground biomass. More than 95% of the standing biomass and litter were removed on these plots. Prescribed burns during plant growth in treatment 2 and 3 were less uniform because of grass growth increased moisture in the fuel. Approximately 20 and 35% of the total fuel load was green in treatments 2 and 3, and only 70 to 80% of the standing biomass and litter were consumed in these treatments. Treatment four had the least uniform burn because approximately 60% of the fuel load was green foliage, and approximately 50% of the total mass was consumed.

Plant density increased about 50% in treatments 1 and 3, 38% in treatment 2, and 8 % in treatment 4 one year after burning (Table 2). Plant density had increased by 32% in T_1 and T_3 and 43% in T_4 compared with the control four years after burning.

FORAGE PRODUCTION

The total annual precipitation was 285 mm in 1985, 479 mm in 1986, 232 mm in 1987, 324 mm in 1988, and 406 mm in 1989. Approximately 60 % of the precipitation occurring in summer an the remaining 49 occurs between October and April. Total live green standing biomass at the end of the growing season increased by 77% (T_1), 83% (T_2), 78% (T_3). But green declined 42% in (T_4). In contrast, current year pro-

Table 2. Mean average (1985-1986) plant density/m² in buffelgrass pastures before and after burning in the northwestern, Mexico.

Treatment	Years after burn 1	2	3	4	Total mean[2]
Prehatching (T_1)	7.3	5.0	4.6	2.5	4.9ab
Hatching (T_2)	9.3	6.3	6.3	5.9	5.6a
2nd & 3rd nymphal instar (T_3)	6.0	6.0	6.1	6.1	5.3a
5th nmphal instar & adults (T_4)	5.8	5.9	5.0	4.5	3.0b
Control (T_5)	3.4	3.0	3.0	2.5	3.0b
Total mean[1]	6.4a	5.2b	5.0b	4.3b	

[1] and [2] Mean within a columm or a row followed by the same letter are not significantly different ($P \leq 0.05$).

duction in the unburned plots was only 30% of the total biomass (Table 3).

Live biomass production declined 48% in the control (Table 3), and is attributed to precipitation and spittlebug damage (nymph and adults). Precipitation effects are important because rainfall was above the long-term average (320 mm) for the two years following the treatments (1985 and 1986) (Martin et al. 1995).

Burning in summer before rapid plant growth and after 50 mm of summer precipitation had less impact on buffelgrass density and forage production than burning before rain or after rapid plant growth (Table 3). During 1985 and 1986 the onset of summer precipitation was later than usual and exposure to intense summer temperatures before summer rains, high soil temperatures and high evaporation may have caused crowns to be damage. Late summer burning, after rapid plants growth, reduced both plant density and productivity. This is attributed to lack of post-burn soil moisture.

Recent-dead biomass accumulated during the four summers after fire (Table 4). Accumulations were greatest in treatment two and three, intermediate in T1 and least in T4. Spittlebug damage in the unburned control was 38% greater than in burned areas.

Old-dead biomass gradually accumulated in the next four summers after burning during summers (Table 5). Accumulations after fire occurs only in summers when rainfall is above the long-term average.

Nymph Spittlebug populations were significantly ($P \leq 0.05$) reduced after burning application four consecutive years (Table 6). Adult spittlebug populations (Table 7) were significantly reduced by prescribed burning buffelgrass prior to spittlebugs 5th instar stage (T_4). Adults found on the prescribed burn plots are attributed to migration from the untreated areas.

Table 3. Mean average (1985-1986) total buffelgrass live-green biomass (kg DM/ha) before and after burning in northwestern, Mexico.

Treatment	Years after burn 1	2	3	4	Total mean[2]
Prehatching (T_1)	1669	1627	1617	1890	1701b
Hatching (T_2)	1662	2896	2624	2537	2429a
2nd & 3rd nymphal instar (T_3)	1364	1664	2579	2763	2092a
5th nmphal instar & adults (T_4)	657	683	689	782	703c
Control (T_5)	1558	1201	746	716	1055b
Total mean[1]	1382b	1614a	1651a	1737a	

[1] and [2] Mean within a columm or a row followed by the same letter are not significantly different ($P \leq 0.05$).

Table 4. Mean average (1985-1986) total buffelgrass recent-dead biomass (kg DM/ha) before and after burning in northwestern, Mexico.

Treatment	Years after burn 1	2	3	4	Total mean[2]
Prehatching (T_1)	0	488	485	605	394b
Hatching (T_2)	0	627	657	713	499ab
2nd & 3rd nymphal instar (T_3)	0	466	825	774	516ab
5th nmphal instar & adults (T_4)	0	341	220	375	234c
Control (T_5)	909	755	572	573	702a
Total mean[1]	181b	535a	552a	608a	

[1] and [2] Mean within a columm or a row followed by the same letter are not significantly different ($P \leq 0.05$).

Table 5. Mean average (1985-1986) total buffelgrass old-dead biomass (kg DM/ha) before and after burning in northwestern, Mexico.

Treatment	Years after burn 1	2	3	4	Total mean[2]
Prehatching (T_1)	0[3]	T	T	T	
Hatching (T_2)	0	T	T	T	
2[nd] & 3[rd] nymphal instar (T_3)	0	T	T	T	
5[th] nymphal instar & Adults (T_4)	0	T	T	T	
Control (T_5)	1168a	1474a	1170a	1097a	1227a

[1] and [2] Mean within a columm or a row followed by the same letter are not significantly different ($P \leq 0.05$).

[3] Old material was less than 20 to 40 g/plots (T= trace).

Table 6. Mean average (1985-1986) spittlebug nymphs densities/m² as influenced by summer burning on buffelgrass pastures in northwestern, Mexico.

Treatment	One year before burning	Years after burn 1	2	3	4	Total mean[2]
Prehatching (T_1)	42	0	0	0	0	
Hatching (T_2)	45	0	0	0	0	
2nd & 3rd nymphal instar (T_3)	35	0	0	0	0	
5[th] nymphal instar & adults (T_4)	55	0	0	0		0
Control (T_5)	43	25b	18c	15c	20b	20

[1] and [2] Means within the control followed by the same letter are not significantly different ($P \leq 0.05$).

Table 7.- Mean average (1985-1986) spittlebug adults densities/m² as influenced by summer burning of buffelgrass pastures in northwestern, Mexico.

Burning treatment	One year before burning	Years after burn 1	2	3	4	Total mean[2]
Prehatching (T_1)	20	1	0	0	0	0.3b
Hatching (T_2)	35	1	0	0	0	0.3b
2[nd] & 3[rd] nymphal instar (T_3)	30	0	0	0	0	0b
5th nymphal instar	36	15	6	28	0	10b & adults (T_4)
Control (T_5)	30		29	25	30	25 22a
Total Mean[1]	30	22b	16b	29a	25ab	

[1] and [2] Mean among rows and columns followed by the same letter are not significantly different ($P \leq 0.05$).

MANAGEMENT IMPLICATIONS

Prescribed burning disrupted the spittlebug cycle, destroyed eggs at the soil surface and removed the litter cover for nymphal stages by removing standing dead biomass.

Prescribed burns applied before the third and fourth development stage significantly reduced spittlebug populations. However, the recomended prescribed burn is that which has the least negative impact on buffelgrass production but the greatest impact in the spittlebug populations. Burning should be applied after accumlative summer precipitation exceeded 50 mm and before active plant growth.

Summer burning is a practical way to control spittlebug populations in buffelgrass pastures, but knowledge of the life cycle of the insect and when it

can be controled without damaging buffelgrass productivity are a key to developed a management option to control the insect in the Sonoran Desert.

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Fire and the Malpai Borderlands Group: One Rancher's Perspective

Bill McDonald[1]

Abstract.—The Malpai Borderlands Group is a landowner-driven organization in southeastern Arizona and southwestern New Mexico. Focusing on the use of managed fire to control brush invasion, it works with state and federal agencies to practice ecosystem management. Concerted cooperative effort and demonstration burns have helped shift agency fire policy in the area from all-out suppression to controlled use.

The Malpai Borderlands Group is a grassroots, landowner-driven organization attempting to practice ecosystem management in the borderlands of southeastern Arizona and southwestern New Mexico. The land is a patchwork of private, state trust and federal land (both BLM and USFS).

The group began as a small discussion group of local landowners and a group of folks who could broadly be described as environmentalists who did not own land in the area, but were concerned about the land. Also involved was a scientist, Ray Turner of the Desert Laboratory in Tucson. Our group came to consensus on two points of concern: one, the biggest immediate threat to our land area was encroaching land fragmentation from subdivision and two, that the greatest long term threat to the productivity and biodiversity of the land (in the absence of subdivision) is encroaching site domination by woody species in former grassland. Dr. Turner felt that the change in the fire regime has occurred as a result of man's activities in reducing natural fuels and in outright suppression of fires. This is probably a major factor in the shift toward woody species dominance that has occurred in this century. Many in our group were inclined to agree with him.

It was fire, one fire in particular, which moved the group into action. In the summer of 1992, a small fire started adjacent to the dirt road which is Geronimo Trail, going east out of Douglas, Arizona through the San Bernardino Valley and the Peloncillo Mountains into New Mexico. The fire was burning dry three-awn grass interspersed with creosote brush and bounded on all sides by thick patches of creosote brush and by Geronimo Trail. There was no chance that the fire would spread outside the confines of the acreage with the three-awn grass. Nonetheless, a suppression crew from the Forest Service showed up to fight the fire. The crew was under contract with the State Land Department of Arizona to do fire suppression work on State Trust land, and suppress they would. The manager of the ranch which held the grazing lease on the state land and owned the interspersed and adjoining private land, begged the fire crew chief to allow the fire to burn the grass, but to no avail. Taxpayer dollars were spent to put out a fire which was going nowhere, which did no harm and had the potential to do good by impacting the creosote brush, reinvigorating the grass and putting organic material in the soil.

The absurdity of the situation moved the ranching community in the area into action. A meeting was called at the Malpai Ranch of Warner and Wendy Glenn, and out of that meeting came two things: a call to the land management agencies to enter into a fire management plan with the ranchers and the creation of a fire map by the ranchers which showed the ranch boundaries, owners names and telephone numbers, and their personal preferences for how they wish a fire start on their ranch to be handled.

A follow-up meeting with the agencies and some of the ranchers resulted in a commitment to an eco-

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system approach to land stewardship and eventually the emergence of the Malpai Borderlands Group as a nonprofit 501(c)(3) organization.

One of the first actions of the Malpai Borderlands Group was to focus on managed fire as a relatively inexpensive and gradual way to control brush invasion and to care for the land. Many in the group felt as though the accepted practice of lighting prescribed burns in the cool season was not adequate to achieve the needed impact on woody species. Tom Swetnam has discovered from tree ring studies that the widespread fires of past ended in the period from 1870-1910 when larger numbers of livestock were brought into the Southwest[2]. This was followed by large numbers of people who built structures that needed protection from fire. Ironically, the remaining open spaces where cattle still graze, are the only places where fire can now be allowed to burn. Ray Turner feels that sustainable, profitable cattle operations may be our best hope in controlling the sprawl of subdivision and the resulting all out suppression of fire[3]. It became the intention of our group of ranchers, scientists and other stakeholders to try to use fire during the season when fires would naturally occur.

The summer of 1993 had seen some 23,000 acres burn in our area, mostly on private land on the huge Gray Ranch in New Mexico. Already, the agencies were responding by taking a more cost-effective approach to suppressing the fires, using natural barriers and backfires as suppression tools.

In 1994, we concentrated on remote Baker Canyon for our first prescribed burn. The targeted area of 12,000 acres was purposely chosen partially because it was so challenging. The principal private landowner had rested the area from grazing in an attempt to build up fire fuels for a burn, but a number of bureaucratic hurdles needed to be crossed. Involved were 3 private landowners, BLM land in two districts in two states, U.S. Forest Service land, and Arizona and New Mexico state trust land. The area fell within the boundaries of two different conservation districts, one in Arizona and one in New Mexico. Some 60 species of concern were identified as possibly inhabiting the canyon, so the U.S. Fish and Wildlife offices in both states were consulted as well as state Game and Fish offices in both states. Mexico lay on the southern boundary of the burn and needed to be consulted as well. Part of the area lay within a Wilderness Study Area and the National Environmental Protection Act and the federal Antiquities Act needed to be followed.

A less determined group might have given up. Through a Herculean effort and after an ongoing parade of biologists, fire experts and others through the canyon, the burn was authorized to be lit in early June. Then another problem arose. It was extremely dry in 1994. Fire crews and equipment were already engaged in suppression activities in what eventually became the biggest fire season in many years and couldn't get loose to ignite and monitor the fire. Then a freak rainstorm fell in Baker Canyon, causing the grass to green up and make the prescription goals unreachable in 1994. Something interesting happened next. Lightning ignited a fire late one afternoon in a canyon to the north of Baker. The Sycamore Canyon burn almost didn't happen. The Safford BLM District, on whose land the fire started, dutifully sent an air tanker to dump water on the fire just before sunset. It was their intention to initiate all out suppression the next morning. At the time, the fire had burned about 150 acres. The landowner who held the BLM grazing lease didn't want the fire suppressed. Working with Larry Allen of the Forest Service, who worked with the Safford BLM District on the Baker Canyon prescription, BLM decided to move to a contain and control strategy and allow the Forest Service primary responsibility for monitoring the fire. The Sycamore fire burned for three days and impacted about 2500 acres. Before the end of the '94 season, approximately 70,000 acres burned in the area, again much of it on the Gray Ranch. The suppression costs were a fraction of what was spent in most of the rest of the country. Without the existence of the Malpai Borderlands Group and the work done in Baker Canyon, it's extremely doubtful such an enlightened suppression response would have been taken. For instance, the Sycamore Fire was the first time in the history of the Safford BLM District that they had not responded to a fire start with all out suppression. The Safford BLM and Coronado Forest have now signed a MOU giving the Forest Service primary fire responsibility for the isolated tract of BLM land which lies within what is primarily U.S. Forest Service land in the Peloncillos.

In the spring of 1995, the infamous three-awn grass, creosote brush acreage along Geronimo Trail

[2] *Tom Swetnam, Presentation on Tree Ring Studies, Animas, NM, 1994.*

[3] *Ray Turner, Letter to the Editor, Arizona Daily Star, Tucson, Arizona, 1996.*

caught fire again. This time it was allowed to burn itself out. In July, conditions in Baker Canyon reached prescription again, and the burn was fully ignited. The primary area was blacklined by air first, and then ignited from the interior. Ignition was primarily aerial for safety and economic reasons due to the rough terrain. The fire burned 6,000 acres and was considered a success. Certainly the cooperation was inspiring. The ignition and monitoring involved the Forest Service, the Safford BLM and the Las Cruces BLM, the Natural Resources Conservation Service, the U.S. Fish and Wildlife Service and a private landowner.

Because it was a prescribed burn, several surveys were done in advance of ignition and monitoring programs are underway and will be ongoing to assess the burn's various impacts. These include vegetative monitoring and studies on the effects on birds and reptiles and amphibians.

On the Gray Ranch, several of the fires burned through monitoring plots which were already in place, Dr. Turner is carrying on a photo monitoring study of the fire effects on the Gray Ranch and also on the Sycamore Burn. Peter Warren of The Nature Conservancy recently completed a survey of agave survival rates on the Sycamore Burn which indicates there was no significant impact to this important food source for nectar feeding bats. Most of these fires have left a mosaic of burned and unburned ground on the same sites due to the rough terrain. This will make comparative studies relatively easy.

The prescribed burn planned for 1996 is the Maverick which will encompass private, State Trust and Forest land, again in both states. Because all the affected private landowners are "fire friendly," a minimum of blacklining will be needed. Because of the success of the complicated Baker Burn, so far the bureaucratic "hassles" have been few. Again an aerial ignition is planned. A weather station is in place to help pinpoint the optimum time for ignition.

The Malpai Group is moving ahead with our coordinator from the Forest Service, Larry Allen, on an overall burn plan for our 800,000 acre area. The Peloncillos are the first leg of the strategic plan.

Our group is also working with the Rocky Mountain Research Station of the Forest Service in setting up several experimental plots which will compare different methods of brush management, including burning.

We are updating our fire map on a Geographic Information System at the Gray Ranch and have been working closely with agencies, landowners and local fire departments to familiarize them with it and help them use it in their decision making.

Fire remains a controversial subject within and without our area. How much fire, when, and how often continues to be debated by ranchers, scientists and government personnel. We are taking the long view and we hope our activities will help to shed some light on the best use of fire for the long term health of our land.

Prescribed Burning in Southwestern Ponderosa Pine

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Abstract.—Prescribed burning is an effective way of restoring the fire process to ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) ecosystems of the Southwest. If used judiciously, fire can provide valuable effects for hazard reduction, natural regeneration, thinning, vegetation revitalization, and in general, better forest health. Relatively short burning intervals are required to maintain the benefits of fire.

INTRODUCTION

Prescribed burning in the United States has been a long tradition, especially in long-needled pine ecosystems. This mimicking of naturally occurring fire is increasingly being used as a silvicultural tool in many western and southeastern forests for restoration of natural processes. Fire stimulates the physical, chemical, and biological environment of forest types and thereby improves health, diversity, and tolerance to disturbances. To sustain valuable recreational, wildlife, as well as commercial products, fuels are reduced systematically by fire to lessen the possibility of extensive replacement of forest communities by inevitable fire. In so doing, other valuable benefits are achieved that have been seriously degraded when naturally occurring fires were replaced by human activities.

Naturally occurring fires will likely never play a role in most ecosystems again because of the changes that have occurred with Euro-American settlement. Therefore prescribed burning should be a highly considered alternative that allows for benefits to be achieved in fire adaptive ecosystems. Developing priorities becomes a crucial part of a prescribed burning program.

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FIRE IMPACTS ON ORGANIC MATTER

Because of changes associated with settlement of the Southwest, ponderosa pine forests developed a cluttered, unhealthy nature, as evidenced by unusually high volumes of dead and living biomass. Instead of having open vistas reported by early explorers and settlers, these forests have become dense and unproductive. As Euro-American settlement increased, the desire to eliminate fire has resulted in a steady increase in flammable organic material.

Almost two decades ago, average loadings of naturally occurring fuels in southwestern ponderosa pine stands were more than 22 tons per acre (Sackett 1979). Loadings from the 62 stands surveyed ranged from 8 to 48 tons. Harrington (1982) verified the heavy fuel loadings from southeastern Arizona, with an average of 34 tons per acre.

The hazard these large amounts of forest floor material represent, because of their burnability, strongly suggest the need for reduction, not to mention other associated problems. Trees of all sizes in these forests have generally poor vigor and reduced growth rates (Cooper 1960, Covington and Moore 1994, Sutherland 1989, Weaver 1951). This condition is likely due to reduced availability of soil moisture caused by intense competition, and by moisture retention in the thick forest floors (Clary and Ffolliott 1969, Harrington 1991). The thick forest floor also indicates that soil nutrients, especially nitrogen, may be limited because they are bound in unavailable forms (Covington and Sackett 1984, Covington and Sackett 1986, Covington and Sackett 1990, Covington and Sackett 1992).

Some of the earliest prescribed fires in the Southwest were on the Fort Apache Indian Reservation in Arizona with about 3,000 acres being burned in the late 1940's (Kallender 1969). From 1950 to 1970, over 300,000 acres were burned, primarily for hazard reduction. The effectiveness of the Fort Apacheburning program in reducing size and severity of subsequent wildfire has been documented (Biswell and others 1973, Knorr 1963). Cool, dry, late fall conditions were the basis for initial fuel reduction burns to moderate fire behavior. Fires were strategically ignited, then allowed to burn over vast acreages. In a series of three burns in 1950, this procedure was used to burn portions of 65,000 acres (Weaver 1952). Forest floor fuel loadings were reduced by 55 percent and dead woody material was reduced by 64 to 80 percent. The effects of this burning operation were evident the following year by a dramatic reduction in thenumber of wildfires and in the acreage burned (Weaver 1952).

Later, in November 1956, a large scale burn was conducted on the Fort Apache Indian Reservation, under cool, clear days with moderately high drought index and low rate-of-spread index (Lindenmuth 1960). Fires were allowed to spread unchecked for 33 days within prescribed boundaries. Under these burning conditions, fuel reduction on 75 percent of the area was deemed unsatisfactory because the reduction was minimal.

In another burn in central Arizona, two distinctly different sites were burned under similar fuel moisture and weather conditions. One site had 75 percent more fuel, and 85 percent greater overstory basal area (Davis and others 1968) than the other site. Consumption was greatest on the site with the most fuel, but 2 years later the net fuel change, including consumption by fire and subsequent litter accumulation, was greater on the site with less initial fuel (37 percent reduction) compared to the heavy fuel site (23 percent reduction). The indication is that more damage was done on the site of higher fuel loading resulting in greater post burn fuel accumulation. This indicates that prescribed burning should be a continuing process, not a one-time event.

In 1976, a research study was initiated in north-central Arizona to investigate the use of interval prescribed burning (repeated burns at specified intervals) to reduce and maintain low hazard fuel conditions (Sackett 1980). On the Fort Valley Experimental Forest near Flagstaff, Arizona, research plots were burned in November to start an interval burning study that includes six different burning rotations. Because of warm, dry conditions that fall, the initial burn was conducted at night. Forest floor material was reduced by more than 60 percent and large, woody material by 70 percent. Consumption was greatest where forest floors were deepest, mostly around large mature trees.

One year later at Long Valley Experimental Forest, a companion study was set up to investigate rotational prescribed burning on another soil type. After a very wet summer, the initial burns at Long Valley were ignited during the day. Even though litter moisture was similar in the two burns, humus moisture was 10 to 15 percent higher at Long Valley; therefore, consumption was considerably less (40 percent). Both study areas had about 15 tons per acre of forest floor material before burning.

In a series of summer burns in southeastern Arizona, Harrington (1987) found that humus moisture and preburn forest floor depth were positively correlated with percent of forest floor reduction.

Additional research burns were conducted at Fort Valley within stands of contrasting structure in 1982 (Covington and Sackett 1992) under low humidities (15 to 24 percent) and moderate air temperature (52° to 62°F). Surface fuel moisture ranged from 7 to 10 percent, and humus moisture ranged from 12 to 20 percent. In stagnated sapling stands (doghair thickets), about 34 percent of the 12 tons per acre loading was consumed, 52 percent of the 16 tons per acre in the pole stands was consumed, and 89 percent of the 55 tons per acre was burned in the mature yellow pine stands. Again, like Harrington's (1987) prediction equation points out, there is a high positive correlation between percent forest floor consumed and preburn loading.

Prescribed burning in southwestern ponderosa pine can greatly, but only temporarily, reduce fuel hazard (Harrington 1981, Sackett 1980); 0.6 to 1.8 tons per acre of needle litter can be cast annually (Davis and others 1968, Sackett 1980). Easily ignitable litter fuels accumulate rapidly to hazardous levels after initial fuel reduction burns. Continual reburns are essential to remove the accumulation and to generally maintain low fuel hazard (Harrington 1981, Sackett 1980). Consumption of the litter layer lessens ignitability and rate-of-spread potential. As more duff, ladder fuels, and large logs are consumed, a reduction in potential fire intensity, total energy release, and resistance to control are realized.

FIRE IMPACTS ON THE STAND

Extensive structural and compositional changes have beset ponderosa pine forests of the Southwest over the past century and a half. Numerous references document the open, park-like nature of presettlement pine stands (Biswell and others 1973, Brown and Davis 1973, Cooper 1960, Covington and Moore 1994). Fires were a regular event of these forests, burning the light surface fuels at intervals usually averaging less than 10 years and as often as 2 years (Dieterich 1980, Weaver 1951). Change began during extensive livestock grazing in the late 1900's (Faulk 1970). As grazing intensified, herbaceous vegetation could not respond, and its coverage greatly declined. This decline led to two subsequent changes: reduced fire spread because of the decrease in fine fuel, and an eventual increase in ponderosa pine regeneration because of reduced competition and fire mortality, and more mineral soil seedbeds (Cooper 1960). Early forestry practices, including fire suppression, further reduced the spread of inevitable fires, and contributed to unprecedented fuel accumulations, and stagnation of seedling and sapling thickets.

Dense sapling thickets which were uncharacteristic to the presettlement ponderosa pine ecosystems are today well represented. Sackett (1980) and Harrington (1982) reported typical stands averaging 2,000 trees per acre, of which 65 percent were 1 to 4 inches d.b.h.. Mechanical thinning has been a tool of choice to improve stand structure and health, but as human and monetary resources become more scarce, intensive treatments are rapidly eliminated.

During previous centuries, fire was the natural thinning agent that kept southwestern ponderosa pine forests open (Cooper 1960). Today, fire can still be used effectively to reduce overstory competition and keep natural regeneration from exceeding the space available for healthy growth. Using fire in current forest conditions is much different than fire-thinning that occurred naturally years ago. Stagnated thickets generally consist of 65- to 80-year old trees, which contrasts to the presettlement fire-thinning of mostly less than 20-year-old trees. In addition, even though trees are small in diameter, they possess thick bark. Instead of being able to girdle or scorch a young, short, thin-barked tree, excessive heat concentrations in the crown are required to effectively kill these taller, thicker-barked trees.

Some investigators have dealt with the use of fire as a thinning tool, and it has been viewed with mixed feelings. Tests on the Fort Apache Indian Reservation showed that, though thinning was spotty from fall fires, the fire did a "reasonably effective and conservative job of thinning" (Weaver 1952). Gaines and others (1958) wrote a supplement to Weaver's report providing data that more or less supported his previous observations. Their conclusions were not as optimistic because of the injury to the commercial overstory. Wooldridge and Weaver (1965), reporting on a prescribed fire designed to thin dense sapling stands on the Colville Indian Reservation, concluded that prescribed fire was a rough and largely unpredictable thinning tool.

Ffolliott and others (1977) reported an effective thinning response from an experimental prescribed fire near Flagstaff, Arizona. They concluded, however, that basal area was not reduced enough for optimal growth of the residual stand. As stated before, one fire does not correct problems associated with more than 100 years of fire exclusion, especially as it applies to thinning.

Using fire to reduce logging slash after a shelterwood cut on the Apache National Forest, Buck (1971) observed considerable overstory mortality. Although the prescribed fires were not designed as a thinning tool, they did accomplish some effective thinning from below. The losses (83 percent) were in suppressed and intermediate trees.

Harrington (1981) reported tree density reductions of small and suppressed tree classes to be 24 percent, 56 percent, and 43 percent on three summer prescribed burns with preburn stand densities of about 2,000 trees per acre.

On the Fort Valley initial prescribed burn mentioned earlier (Sackett 1980), stagnated reproduction and sapling stems were reduced from an average of 1,553 to 912 per acre. In the companion study at Long Valley, fire intensity and fuel consumption were lower because of wetter fuel conditions. An average of only 180 stems per acre were killed by the fires in the reproduction/sapling size classes. Virtually none of the small poles were killed outright by the fire.

No known studies or reports deal specifically with the problem of developing and using definitive burning techniques for thinning. Most references deal with a single fire as an answer to the problem. Experiences of studies located on the Fort Valley and Long Valley Experimental Forests near Flagstaff,

Arizona,and study areas in southwestern Colorado and southern Arizona, support Cooper's (1961) suggestion that quality of fuel as well as quantity is essential for producing controlled high intensity fires in dense stands.

Repeat burning in higher quality and quantity fuel does a better job of thinning stagnated stands than single burning in thick forest floors.Work in surface fuel characteristics and experience with many prescribed fires indicate that only the newly cast needles (L layer) and upper portion of the fermentation layer (F) actually burn as flaming combustion in heavy, old forest floor accumulations. The lower F layer is matted and bound tightly together by mycelium hyphae as is the H layer below it. As a result, the lower portion of the F layer acts more like a solid piece of fuel rather than as individual particles as in the L layer, and does not burn well.

In an undisturbed, well-developed forest floor, newly cast needles become rapidly colonized and bound by mycelium and therefore less burnable. When fire spreads over the forest floor, most of the fungi are destroyed. Needles that fall after a fire do not become readily infected and a much deeper layer of pure litter accumulates. When fire is applied a second time, all material cast since the initial fire is consumed (up to 3 or 4 tons per acre). Fire intensity, rate-of-spread, and flame length are much higher in response to the greatly increased available fuel.

Crown scorch and consumption are more effective mechanisms for killing trees and thinning stands than bole girdling. Many of the stagnated sapling stands arose from the famous 1913 and 1918 seed crop and subsequent regeneration. Although the trees have grown little in diameter and tree height, bark thickness has progressed normally through the past 80 years. The unusually thick bark prevents heat of low intensity fires from penetrating enough to kill trees. Subsequent burns in deep litter result in high intensity fires which cause extensive crown damage yet little cambium damage.

The most critical element in the use of fire as a thinning tool is the burner's ability to manipulate the fire or the fire environment or both to achieve slow-dissipating, high-temperature air in the crowns. Fire can be manipulated in a number of ways. Adjusting the direction of fire spread relative to wind direction is the most common technique. Heading or uphill fires move at a speed commensurate with windspeed, creating longer flame lengths, greater speed, and higher intensities than backing fires which move against the wind (or down hill) and progress very slowly with short flame lengths and low intensities. Backing fires seldom thin stands.

FIRE IMPACTS ON PONDEROSA PINE REGENERATION

Very little regeneration of ponderosa pine has taken place in the last 30 to 40 years in untreated stands because of either dense overstory, thick forest floor, or herbaceous competition. To maintain a multi-aged character, pine regeneration is desired in places where the overstory is gone from stress related mortality, harvesting, or fire mortality. Ponderosa pine is a difficult species to regenerate in the Southwest primarily because regular periods of moisture stress are caused by droughts and competition from grasses early in the growing season (Larson and Schubert 1969a, Pearson 1950). Numerous papers point out the difficulties encountered with planting, seeding, and natural regeneration (Heidmann and others 1982, Larson and Schubert 1969b, Rietveld and Heidmann 1974). Prescribed burning is valuable for increasing the probability of obtaining natural regeneration, especially on the silty, volcanic soils of northern Arizona.

Soil moisture seems to be the most critical factor in seedling establishment. Therefore, any activity that results in an increase in available moisture or an increase in soil volume tapped for moisture by roots would be beneficial. Mineral soil with a light litter covering is considered the optimum seedbed (Pearson 1950, Schubert 1974), because it allows best seed and seedling contact with available moisture. Much precipitation can be absorbed by a deep forest floor and then lost through evaporation without reaching the root zone (Clary and Ffolliott 1969).

Much research and observation have shown the beneficial effect of forest floor reduction on pine establishment. Harrington and Kelsey (1979) illustrated the deleterious effect of a deep organic layer and competing vegetation on ponderosa pine establishment in Montana. An additional finding was the much greater size of pine seedling crowns and roots in burned plots, presumably from an increase in available nitrogen. Pearson (1923) noted long ago that spots where slash piles burned produced large numbers of rapidly growing pine seedlings. Reduc-

tion of grass competition was the suggested benefit. Reports by Weaver (1952) and Ffolliott and others (1977) showed much greater pine seedling establishment on burned than unburned seedbeds. Heidmann and others (1982) studied sites of best natural ponderosa regeneration in a harvested watershed in central Arizona. Seventy percent of the sites adequately stocked, had been burned prior to a moderate cone crop.

As part of the fire research at Fort Valley Experimental Forest, burned and unburned seedbeds were surveyed after the 1976 seed crop (Sackett 1984). Burned plots had 2,600 seedlings per acre compared with 833 seedlings per acre on unburned control plots. Two years later, the burned plots still supported over 500 seedlings per acre, while no seedlings were found on the control plots. In a companion study, seeds falling on an undisturbed forest floor seldom reached mineral soil (Haase 1981). Sackett (1984) showed a high correlation ($r^2 = 0.92$) between area of fire-exposed mineral soil (square feet) and amount of stocking: 83 percent of the new pine seedlings germinated on microsites where the forest floor was partially or totally consumed by fire. Another confirmation of this benefit came from a prescribed burning study in southwestern Colorado (Harrington 1985), where 20 times more pine seedlings per acre were located on burned units than on units with unburned forest floors and Gambel oak competition.

A 1984 pine seedling survey at Fort Valley Experimental Forest revealed a more pronounced regeneration success as a result of burning. In 1983, seeds were cast at a rate possibly rivaling the record year of 1918. By summer, 1984, the burned seedbeds averaged over 90,000 seedlings per acre, while the unburned plots had 26,000 seedlings per acre. In fall 1984, two of the three previously burned plots were reburned as part of the burning rotation study. One plot had 4 years of litter accumulation and the other had 8 years of accumulation. Four years after this burn, the following seedling distribution was found: all seedlings were killed on the plots burned with 8 years of litter, 7,800 seedlings per acre remained on the plots burned with 4 years of litter, 15,000 seedlings per acre remained on the plots burned before seed fall, and 1,200 seedlings an acre remained on the controls.

Over the 18-year course of burning at Long Valley Experimental Forest, natural regeneration was never as pronounced as at Fort Valley. Before burning in the fall of 1995, however, seedlings were surveyed on annual, biennial, and 6-year rotation burn plots, and averages were 39,900, 9,500, and 35,200 seedlings per acre, respectively.

It is interesting to note that even the oldest seedbed (time since last burned) had excellent regeneration. Even though a 6-year litter accumulation is 3 to 4 tons per acre, it is in a form that (1) seeds can drop through to mineral soil, and (2) the needles act as a mulch against evaporation. Obviously many of these seedlings were consumed by the next fire, but for managers who would like to promote and encourage the survival of seedlings, protecting them from fire by simply not igniting in those areas would be an easy matter.

It is difficult to define specific forest floor moisture content conditions under which an optimum seedbed will result from burning because of the variable pattern of consumption. Generally, in dense, or otherwise fully stocked groups within stands, prescribed burns will create few mineral seedbeds. However, on sites where mature trees have been or will be removed, fires burn to mineral soil within a large range of forest floor moisture contents. Places that do not need seedling regeneration typically will not have much mineral soil exposed, and the places where pine regeneration is desired generally have mineral soil exposed by fire.

FIRE IMPACTS ON SOIL NITROGEN

Fire also serves as a recycling agent for nutrients that are organically bound in the forest floor, woody material, and herbaceous vegetation. During the burning process, organic nutrients may be oxidized to inorganic forms that are readily available for use by trees and herbaceous vegetation. A portion of some nutrients is volatilized and lost from the site. Loss of nitrogen (N), which has a relatively low volatilization temperature, is especially important because it is usually the most limiting plant nutrient in forest ecosystems (Maars and others 1983).

Burning at Fort Valley has produced increases in inorganic nitrogen in the soil (Covington and Sackett 1992). Concentrations of ammonium-N in mineral soil are directly related to the amount of material consumed. As with other aspects of prescribed burning, repeat prescribed burning had additional soil nutrient impacts. Inorganic N levels are short-lived

without repeat burns (Covington and Sackett 1986). Four years after an initial burn, concentrations of ammonium-N approached control levels, but were elevated again following additional burns.

These fire induced increases in inorganic nitrogen probably contribute to the increased seedling establishment of both herbaceous vegetation (Vose and White 1987) and ponderosa pine (Sackett 1984), as well as higher herbaceous biomass production and greater foliar nitrogen concentrations (Harris and Covington 1983, Oswald and Covington 1984, Andariese and Covington 1986, Covington and Sackett 1990) and increased pine seedling growth (Owen 1985). Increased growth was further documented at Fort Valley where 6-year-old pine seedling heights approached 24 inches on burned plots in contrast to 6- to 8-inch heights for the same-age seedlings on unburned controls.

Diameter growth of pole-size trees was measured in 1988 at Fort Valley (Peterson and others 1994). Moderate changes in diameter growth occurred after 1984, 8 years after the largest increases of inorganic nitrogen occurred. Trees in the 1-, 2-, 8-, and 10-year rotation burns had slower growth than controls after 1984, presumably because of moderate fire stress. The 4- and 6-year rotation burns had slightly higher growth rates after 1984. Lack of large growth increases is likely due to the continued dense stocking.

FIRE PRESCRIPTIONS

Much has been learned within the last 50 years about the use of fire in the Southwest. Many fire experts have developed their skills primarily through personal experience, learning from failures as well as successes. This type of knowledge is difficult to pass on to less experienced individuals. However, there is now enough documentation of research and operational burns to provide general guidance for fire prescription and effects. Unique combinations of stand, fuels, vegetation, and terrain may preclude the use of the following prescriptions and effects information. Therefore, we recommend a thorough assessment of site characteristics. A generalized set of fire prescription parameters was derived from the prescribed burns discussed earlier.

In forested sites where fire has been absent for decades, the initial fuel reduction burns should be conducted in the fall or early spring when temperatures and humidities are moderate. Fall burning can begin as early as mid-September and can continue in some years into December, or later in snow-free years. However, smoke dispersal is a distinct problem in the typical stable atmosphere of the winter months.

The following prescription parameters are the primary variables that determine whether a fire will burn successfully, hazardously, or not at all. On sites where reduction of natural fuels is desired, maximum daytime air temperatures should be between 50°F and 75°F. Below 50°F, moderately dry fuels (9 to 12 percent moisture) burn poorly and above 80°F extensive overstory crown scorching is likely. Minimum relative humidities should not drop below 20 percent or exceed 40 percent. Fuels subjected to a series of low humidity days become hazardously dry. Also, very low humidities are frequently accompanied by temperatures above 80°F. If minimum humidity exceeds 40 percent, light surface fuels are generally too moist to burn well. Windspeed at flame height should be between 3 and 8 mph. Slope effects can compensate for lack of wind. A fire burning with little or no wind and no effective slope will not spread well or will cause extensive crown heating if fuels are dry. Windspeeds greater than 10 mph can result in erratic fire behavior. Surface pine needles ideally should contain 5 to 12 percent moisture. Below 5 percent, ignition and rates-of-spread are too rapid, and above 12 percent, burning is patchy and incomplete with slow rates-of-spread.

Caution must be used since not all combinations within the range of temperatures, humidities, windspeeds, and fuel moistures described above, are safe and effective. For example, if burning conditions are approaching the upper temperature and windspeed limits and the lower humidity and fuel moisture limits, a very intense, rapidly spreading fire will result. However, experienced burners can use the upper limits of one parameter to make up for a deficiency in another. For example, a combination which provides good burning conditions is low humidity (15 to 20 percent) and low temperatures (40 °F to 50 °F). These situations do occur in late fall throughout the Southwest.

Because damp, cool fall weather often results in poor burning conditions, summer burning during the monsoon season in the Southwest has been studied as a successful alternative (Harrington 1981, 1987). The amount of drying that follows fuel-saturating

rains will determine fire behavior and fuel consumption. Using the same prescription ranges during the summer rainy season should permit successful fuel reduction burns. More attention to air temperature limits and erratic winds is needed, however.

Maintenance burning is necessary to keep the recurring fuel hazard to a minimum (Davis and others 1968, Gaines and others 1958, Harrington 1981, and Sackett 1980). Since most of the light, fire-created fuels accumulate within 3 years of burning, we recommend a repeat burn within that period. Generally, repeat burns in light, needle fuels are easily managed. Smoke management issues also decrease with maintance burning due to the shorter duration and lower volumes of smoke generated with the less fuel. The window of burning season and ambient conditions is broader than for initial burns, with warmer, drier, windier situations being advantageous to the conduct of the burn (Harrington 1985). Air temperatures should range between 55 oF and 85 oF, humidities from 15 to 40 percent, windspeeds from 5 to 12 mph, and litter moisture from 5 to 10 percent. After the second or third burn, annual litter accumulation should return to a level relative to natural attrition. From this point, burning need only be conducted at intervals of about 7 to 10 years to maintain a low hazard. However,as burning rotations increase, weather and fuel parameters during the burn need to be more moderate.

If a reduction in sprouting shrubs is a major management goal for fuel and competition reduction, then a distinct program of repeat burning is needed. For Gambel oak management, we suggest an initial fuel reduction burn in fall followed by 2 or 3 mid-August burns, 2 years apart (Harrington 1985).

Because health and stability of presettlement southwestern ponderosa pine ecosystems was keyed to frequent fire, importance of proper fire reintroduction seems clear. Prescribed fire, in mimicking the natural role of fire, can be an ideal tool for accomplishing many forest management objectives. The ideas and prescriptions presented are very general, and prescribed burning anywhere is site-specific. Managers must learn how to prescribe conditions that relate specifically to their particular resource objectives.

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Fuel Loadings in Southwestern Ecosystems of the United States

Stephen S. Sackett and Sally M. Haase[1]

Abstract.—Natural forest fuel loadings cause extreme fire behavior during dry, windy weather experienced during most fire seasons in the Southwest. Fire severity is also exacerbated from burning heavy fuels, including heavy humus layers on the forest floor. Ponderosa pine and mixed conifer stands possess more than 21.7 and 44.1 tons per acre of total forest floor fuel, respectively, in Southwestern forest ecosystems.

INTRODUCTION

The ignition, buildup, and behavior of fire depends on fuels more than any other single factor (Brown and Davis 1973). Fuels, made up of the products of photosynthesis, continually accumulate as any ecosystem progresses. In any fire, the rate at which readily available fuels are consumed determines fire intensity (Byram 1959). This dynamic fire behavior has a direct effect on the environment in which it is burning, usually seen in above-ground flora. Once the active flaming front has passed, glowing, smoldering combustion creates totally different effects on the same environment, usually the unseen fire severity effects at or below ground. Fuels and their many characteristics provide the basis for assessing the outcome of fire, in any form, whether it be the above-ground fire intensity effects or the below-ground fire severity effects.

Of the many types of fire that exist in wildland ecosystems, live fuels in the Southwest tend to be the most spectacular when they burn. Chaparral fires, whether in California or Arizona, can spark the attention of even the most passive observer. Fire in live ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) crowns can race over the countryside at breakneck speeds, uncontrollable by even the most aggressive suppression efforts.

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But for all the live fuel fires, fires occurring in long-needle conifer surface and ground fuels in the southwestern United States and northern Mexico have more relevance to protecting and maintaining the health of forested ecosystems. The amount of forest floor fuel has a pronounced effect on fire hazard, moisture relations, forage production, and the general health of coniferous forests.

FACTORS AFFECTING FUEL ACCUMULATION

The structural and compositional changes in southwestern ponderosa pine over the past 100 years or more have been repeatedly documented (Biswell and others 1973, Brown and Davis 1973, Cooper 1960). What was once an open, park-like ecosystem, maintained by frequent, low-intensity fires, is now a crowded, stagnated forest. In addition to stand changes, general fire absence has led to uncharacteristically large accumulations of surface and ground fuels (Kallander 1969).

The natural accumulation of pine needles and woody fuels is exacerbated by the very slow decomposition rates characteristic of the dry, southwestern climate (Harrington and Sackett 1992). Decomposition rate (k) (Jenny and others 1949) is the ratio of steady state forest floor weight to the annual accumulation weight. Harrington and Sackett (1992) determined k values of 0.074, 0.059, and 0.048 for sapling thickets, pole stands, and mature old-growth

groves, respectively. Decomposition rates this slow, and which Olson (1963) considers quite low, border on desert-like conditions. Humid, tropic conditions would have k values approaching 1.0 where decomposition occurs in the same year as the material is dropped on the ground.

Fuel loading estimates can be obtained in a number of ways. Slash loading can be predicted from information obtained in timber sale surveys (Brown and others 1977, Wade 1969, Wendel 1960). Although originally developed to inventory activity-generated downed woody material, Brown's (1974) planar intersect method is oftentimes used by managers to determine natural woody fuel loadings. Forest floor weights, however, have been studied only to a limited extent in Arizona, with conditional success because of inherent variability. Ffolliott and others (1968, 1976, 1977), Aldon (1968), and Clary and Ffolliott (1969) studied forest floor weights in conjunction with water retention on some Arizona watersheds. These and other works included prediction equations relating forest floor weight to stand basal area (Ffolliott and others 1968, 1976, 1977), age (Aldon 1968), height and diameter (Sackett and Haase 1991), and forest floor depth (Harrington 1986, Sackett 1985).

Because of the variable nature of forest floor fuel loading, Sackett (1979) set out to determine the range over the entire Southwest of unmanaged ponderosa pine and mixed conifer stands.

PONDEROSA PINE FUEL LOADINGS

The importance of forest floor loadings has been generally overlooked by managers when accessing total fuel loadings. The forest floor consists of the litter (L) layer, recently cast organic material; fermentation (F) layer, material starting to discolor and break down because of weather and microbial activity; and the humus (H) layer, where decomposition has advanced. The loosely packed L layer and upper portion of the F layer provide the highly combustible surface fuel for flaming combustion and extreme fire behavior during fire weather watches and red flag warnings. This combined layer of fuel could be referred to as the fire intensity (FI) layer of the forest

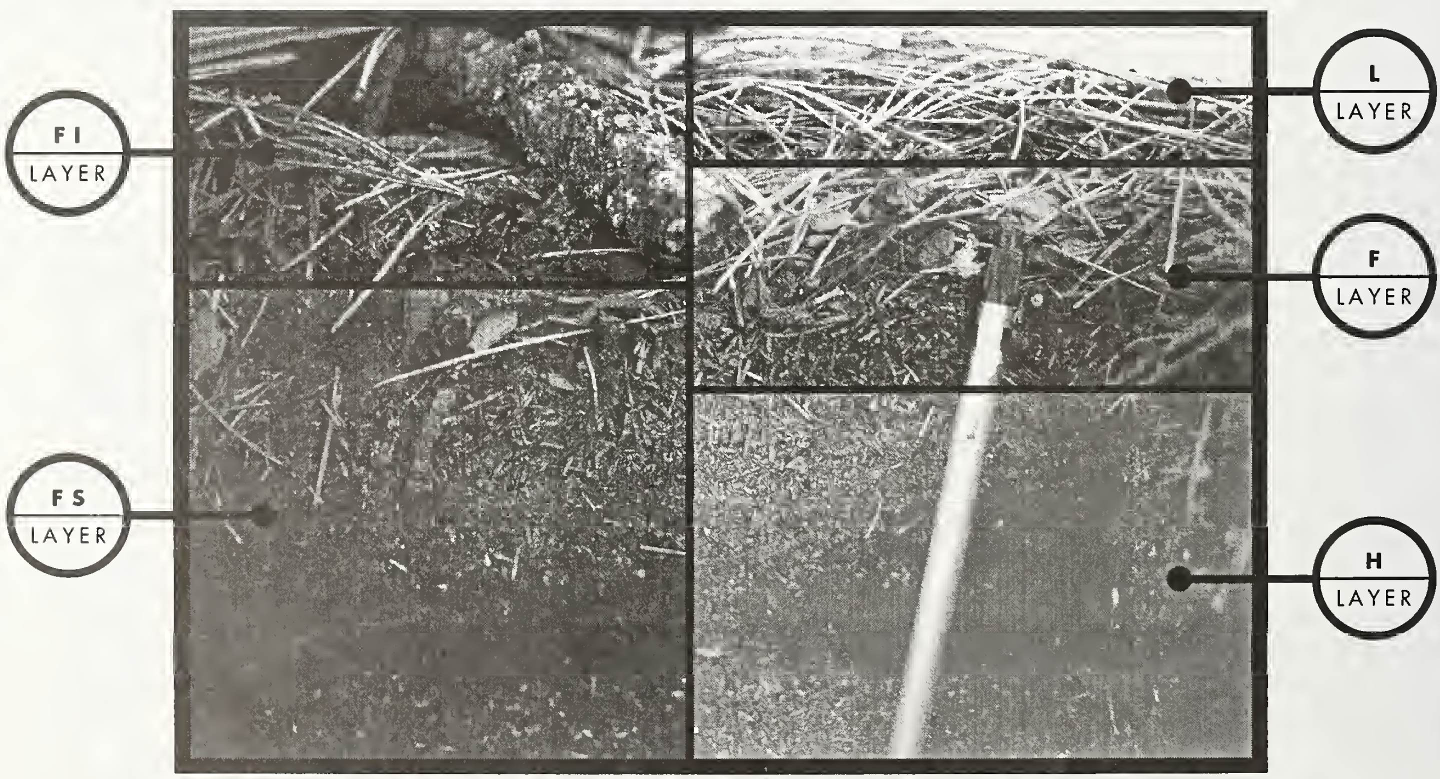

Figure 1. Section of heavy forest floor material indicating fire intensity (FI) layer of fuel and fire severity (FS) layer of fuel in relation to the three layers of forest floor material (L, F, and H).

floor. The lower, more dense part of the F layer and the H layer make up the ground fuel that generally burns as glowing combustion. This combined layer of the forest floor could be referred to as the fire severity layer (FS) (Figure 1).

Sixty-two Case Studies

In the late 1970's, forest floor fuels were sampled in 62 stands in Arizona and New Mexico (Sackett 1979). Throughout the Southwest, unmanaged stands of ponderosa pine had from 4.8 tons per acre in a stand on the Tonto National Forest to more than 20 tons per acre in a stand on the north rim of the Grand Canyon National Park. The next two heaviest weights (18.3 and 18.0 tons per acre) also occurred on the north rim of the Grand Canyon. Mean forest floor loading for the entire 62 stands measured was 12.5 tons per acre. When woody material greater than 1-inch diameter was added, the average increased to 21.7 tons per acre. The heavier material does not have much to do with extreme fire behavior, except as a spotting potential; these fuels do contribute to localized severity when burned. Table 1 provides a summary of forest floor fuel loadings observed in the Southwestern ponderosa pine ecosystem.

Of the 12.5 tons per acre average of forest floor fuel load found in the Southwest, about 1.0 ton per acre was L layer material, 3.8 tons per acre was in the F layer, and 6.1 tons per acre was made up of humus (H layer). Small diameter woody material and other material comprised the remaining 1.8 tons per acre. Large, woody material that accounted for 42 percent of the total fuel load consisted of 1.4 tons per acre of material 1 to 3 inches in diameter, 5.0 tons per acre of rotted, woody material 3 inches or more in diameter, and the remaining 2.8 tons per acre was sound wood greater than 3 inches in diameter.

Stand Conditions Affecting Fuel Loadings

Soon after this extensive fuel survey was conducted, two prescribed burning studies were established; one at Fort Valley Experimental Forest on a basalt soil, the other on the Long Valley Experimental Forest on a sedimentary soil. Both are located near Flagstaff, Arizona. Additional forest floor fuel sampling was done on those areas. At Fort Valley, 15.2 tons per acre of forest floor lay on the site before burning (Sackett 1980). Although more than the Southwest average, it was not much different than that found on the Coconino National Forest (14.7 tons per acre) or the Kaibab National Forest (15.5 tons per acre), both of which were nearby. Only 7.2 tons per acre were found in woody material greater than 1-inch diameter.

At Long Valley, an almost identical forest floor fuel load was measured (15.7 tons per acre). The difference between the sites was the larger amount of large, woody material (16.6 tons per acre) at Long Valley because of some sanitation logging during the 1960's.

Another site at both areas was established to glean more information about initial prescribed burns in heavy forest floor fuels. At that time, a concentrated effort was made to develop an easier method of estimating forest floor fuel loading. Earlier attempts were made by others to define forest floor fuel loading by the depth of forest floor (Eakle and Wagle 1979, Sackett 1979). To date, nearly 900 individual square foot samples have been taken between the two areas to further define forest floor fuel loading and residual fuels after burning. Prediction equations have proven extremely accurate (r^2 = 0.87 to 0.90), but tend to be site specific. For instance, Fort Valley has about 17.7 tons per acre where the forest

Table 1. Ponderosa pine fuels in the southwestern United States (Sackett 1979).

Location	No. of sites	Forest floor and 0–1 inch dia. wood (T/ac.)	Woody fuel >1–inch dia. (T/ac.)	Total fuel (T/ac.)
Kaibab NF	4	15.5	8.6	24.1
Grand Canyon NP	4	17.5	5.6	23.1
Coconino NF	4	14.7	19.8	34.5
Tonto NF	2	6.5	2.7	9.2
Apache-Sitgreave NF	14	11.3	11.2	22.5
San Carlos Apache IR	3	14.4	8.4	22.8
Fort Apache IR	2	15.1	20.5	35.6
Gila NF	10	11.2	7.3	18.5
Navajo IR	1	9.4	4.9	14.3
Cibola NF	3	8.8	8.8	17.6
Santa Fe NF	3	13.2	14.6	27.8
Carson	4	13.3	4.3	17.6
Bandalier NM	1	11.6	3.0	14.6
Lincoln	2	13.9	7.1	21.0
San Juan NF	5	11.9	4.8	16.7

floor is 2 inches thick, whereas Long Valley has 22.2 tons per acre of forest floor fuel where the depth is 2 inches.

Not only is there wide variation from site to site in ponderosa pine ecosystems, but vast differences exist within stands with respect to overstory characteristics. Experience indicates four separate conditions: sapling (doghair) thickets, pole stands, mature, old-growth (yellow pine) groves, and open areas in the groves without crowns overhead. In the two additional study areas, the following forest floor fuel loadings with woody material 1 inch or less, were measured in each of four overstory conditions (Sackett 1985).

Sapling T/ac	Poles T/ac	Open T/ac	Mature T/ac
14.8	18.2	10.4	48.4

Annual rates of forest floor accumulation are also commensurate with overstory conditions. Sapling thickets produce as much as 1.1 tons per acre per year, pole stands 1.5 tons per acre per year, and mature, old-growth groves as much as 2.1 tons per acre per year.

A substantial amount of forest floor material remains after an area is initially burned. This amount persists even with repeat applications of fire. Figure 2 shows the mean amount remaining (old) after burning, and the average accumulation of fuel (new) since the last fire on the various rotation plots of the two long term study areas.

SOUTHWESTERN MIXED-CONIFER FUEL LOADINGS

Southwestern mixed-conifer stands contain varying proportions of Douglas-fir [*Pseudotsuga menziesii*

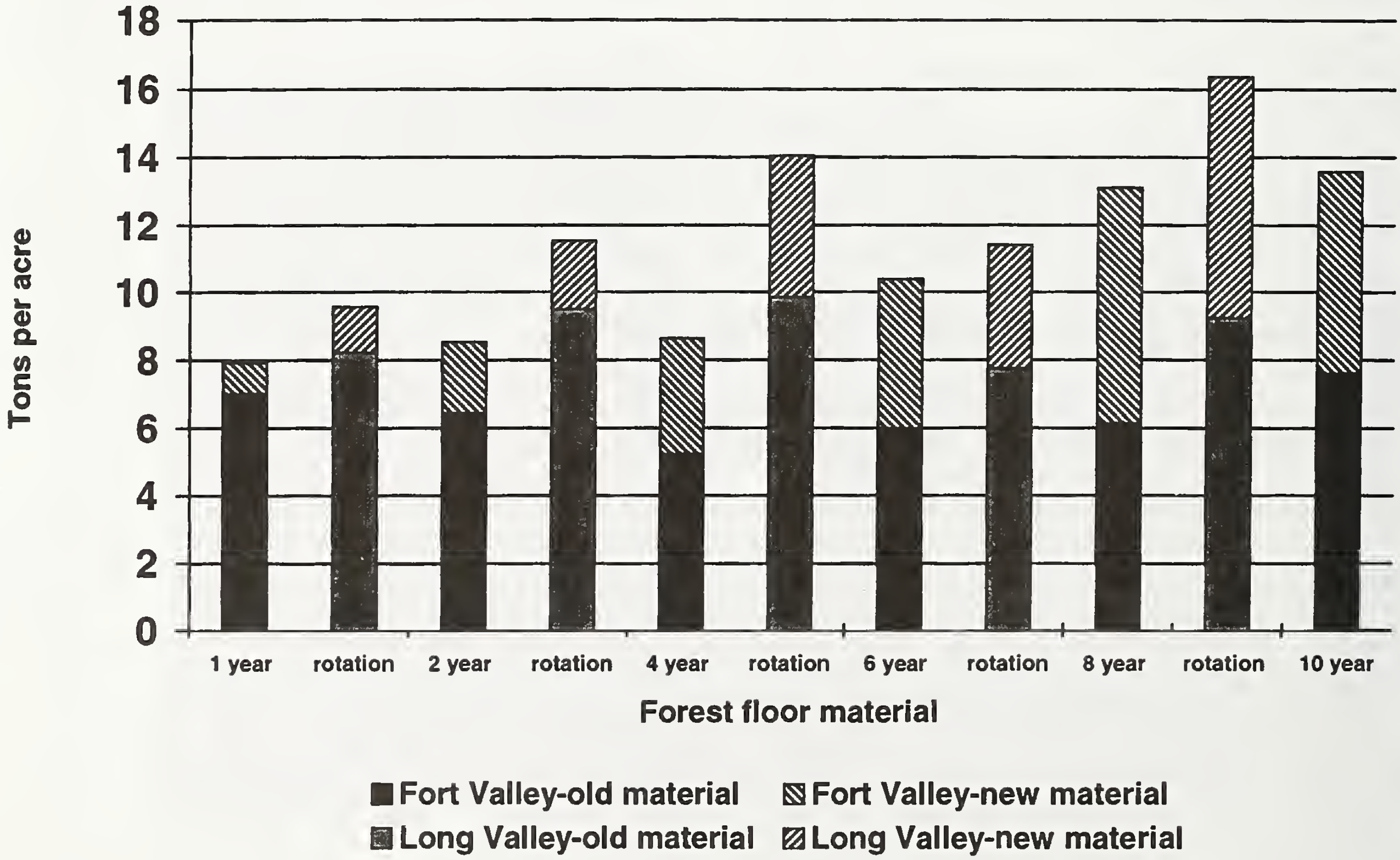

Figure 2. Mean forest floor remaining after prescribed burning (old) and average amount of forest floor accumulating since the last fire (new) for Fort Valley and Long Valley.

(Mirb.) Franco], white fir [*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.], corkbark fir [*Abies lasiocarpa* var. *arizonica* (Merriam) Lemm.], ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.), blue and Engelmann spruce(*Picea pungens* Engelm. and *Picea engelmannii* Parry ex Engelm.), and aspen (*Populus tremuloides* Michx.). Forest floor loadings in 16 stands with varying degrees of species composition were also sampled (Sackett 1979). Mean forest floor fuel loads were more than 22 tons per acre, with a range from 13.6 to more than 30 tons per acre (Table 2). Woody fuels greater than 1-inch diameter, with most of the weight in classes greater than 3-inch diameters, ranged from 9.6 to 58.4 tons per acre and on average added another 21.9 tons per acre to the forest floor fuel average. L and F layers were comparable to ponderosa pine, but the H layer was found to be more than twice (12.3 tons per acre) that of ponderosa pine. In three watersheds located in east-central, Arizona, Ffolliott and others (1977) found an average of 21.1 tons per acre in mixed conifer stands. Gottfried and Ffolliott (1983) found annual litterfall in a southwestern mixed-conifer stand to be about 0.4 ton per acre per year.

Table 2. —Mixed conifer fuels in the southwestern United States (Sackett 1979).

Location	No. of sites	Forest floor and 0–1 inch dia. wood (T/ac.)	Woody fuel >1–inch dia. (T/ac.)	Total fuel (T/ac.)
Apache-Sitgreaves NF	9	26.2	27.0	53.2
Bandalier NM	1	18.7	13.6	32.3
Carson NF	3	20.0	13.7	33.7
Lincoln NF	2	13.4	20.5	33.9
Sante Fe NF	1	13.6	11.4	25.0

SUMMARY

Forest floor fuel loadings and total fuel loadings in southwestern coniferous forests are substantial. Rapid accumulation and extremely slow decomposition of fuels, coupled with extreme fire weather, produce extremely hazardous conditions that produce intense, severe wildfire. Fuel accumulation is site specific, but can be estimated using prediction equations developed for those sites that relate loading with stand basal area, tree height or diameter, or with depth of forest floor. Stand overstory age also influences the rate of accumulation of fuel. It needs to be remembered too, that not all forest floor fuel is consumed in an initial prescribed burn. A logical response is to develop prescribed burning programs that reduce and maintain much lower loadings of forest fuels.

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Prescribed Fire in the Pine Forests of Northwestern Chihuahua

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Abstract.—Studies to determine the feasibility of using prescribed fire to prevent fire in the forests of northwestern Chihuahua were initiated in 1982 at an experimental level. These studies have resulted in valuable information on the importance of prescribed fire in protecting and, at times, enhancing forest resources. It is anticipated that the information obtained from these studies will be useful to the Secretariat of the Environment, Natural Resources and Fish in carrying out their the assigned responsibilities and operations.

INTRODUCTION

Fire has been considered one of the most important factors in the destruction of forest resources, above all when fire occurs at high intensity and frequency. Forest fire is a threat in periods of drought, when there is a build-up of fuels, and where the continuity of vegetative cover allows the fire to spread.

Fire is common in the cold temperate forests of Mexico on sites of high accumulations of natural fuels (needles, small twigs, etc.), residues from forest utilization, and both. These conditions, coupled with the frequently encountered long periods of drought, irresponsible human activities, poor road infrastructure, and ineffective of prevention and suppression techniques, result in high risks of fire occurring with an almost certain regularity. According to Vazquez and Segovia (1989), up to 40 percent of several states in Mexico can be affected by fire in a dry year. In particular, the State of Chihuahua has experienced an exceptionally high number of forest fires in the recent years (SARH 1989).

The use of prescribed fire as a tool to combat the negative effects of wild forest fire is being explored in Mexico at this time. Prescribed fire has been used extensively in many areas of the United States and, to a lesser degree, in the States of Jalisco, Micheocan, Durango, and Chihuahua in Mexico.

Studies to determine the feasibility of using prescribed fire to prevent fire in the forests of northwestern Chihuahua were initiated in 1982 at an experimental level. This paper reports on the knowledge obtained from these studies.

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DESCRIPTION OF STUDIES

An agreement to cooperate in the area of multiple use management of natural resources was established in 1981 between the Instituto Nacional de Investigaciones Forestales, Mexico, and the Rocky Mountain Forest and Range Experiment Station and Southwestern Region of the USDA Forest Service. A particular emphasis of this agreement considered the effects of prescribed fire on the forest ecosystems in northwestern Chihuahua. Therefore, studies were initiated to evaluate the effects of prescribed fire on seedling mortality, soil properties, accumulations of fine fuels, vegetative cover, water flow, and infiltration rate.

Sanchez-Cordova and Dieterich (1983) studied prescribed fire in a stand of mature *Pinus durangensis* trees. Nine study plots 50-by-50 m were established to study the effects of fall burning, winter burning and no burning (a control) on the amounts of fuel, vegetation, and soil nutrients. Prefire fuel accumulations on the plots ranged from 51.1-to-52.5 t/ha. The fall burning in November 1983 was prescribed for conditions of 10°C air temperature, 43% relative humidity, a wind velocity of 8-to-10 km/hr, and a fuel moisture content of 16%. The fire spread at a rate of 33 m/hr. Conditions for the winter burning in February 1983 were a specified air temperature of 6.6-to-

11°C, a relative humidity of 21-to-57%, a wind velocity of 6-to-9 km/hr, and a fuel moisture content of 14-to-15%. The fire spread was 21 m/hr.

The prescribed fire resulted in a higher mortality of *Pinus durangensis* seedlings than observed on the control plots in the first year after burning, with winter burning killing more seedlings than fall burning. Specifically, 87% of the seedlings were killed as a result of the winter burning, 53.6% were killed by fall burning, and a mortality of 19.8% occurred on the control. Sanchez-Cordova and Dieterich (1983) found that prescribed burns did not alter soil chemistry and physical characteristics of the soil on the study plots.

Alanis-Morales and Sanchez-Cordova (1994) reported on the effects of prescribed fire on reducing the amounts of fuels in a stand of immature *Pinus durangensis* trees, and secondary effects of prescribed fire on vegetative succession, survival of residual trees, and attacks of insects. Trees in the stand, 28 years in age, and averaged 12 cm in dbh and 8 m in height. Study plots 30-by-30 m were burned in the winter and fall according to prescribed conditions. Accumulations of heavy fuels on the plots before burning ranged from 11.7-to-44 t/ha, and accumulations of fine fuels ranged from 10-to-31.3 t/ha. Conditions for the winter burning in January 1989 were a specified air temperature of 4°C, a relative humidity of 90% at 1100 hr, and 58% at 1400 hr, a wind velocity of 12 km/hr, and a fuel moisture content of 12.3% The fall burning in October 1989 was prescribed for conditions of 11°C air temperature, 52% relative humidity, a wind velocity of 6.5 km/hr, and a fuel moisture content of 12.6%.

The winter and fall burnings both reduced heavy fuels by about 37.7%. The winter burning reduced the fine fuels by 20.7%, while the fall burning reduced the fine fuels by 24.1%. Alanis-Morales and Sanchez-Cordova (1994) also observed that the pioneering herbaceous plants on the study plots after fire were predominantly *Pteridium aquilinum* and *Lupinus mashallianus*. Only 8 of the total of 1,090 trees 7.5 cm and larger in dbh on the plots were killed as a consequence of burning. Insects had not attacked the residual trees in the stand two years after the prescribed fire.

Alanis-Morales (1995) evaluated the effects of prescribed fire on water flow and infiltration rate within a *Pinus arizonica* stand. Twelve study plots 5-by-20 m were established to study the effects of fall burning in drought conditions, winter burning, and no burning (a control). The fall burning treatment was repeated one year after the initial burning. Thirty-four measurements of infiltration rate were obtained at 3 sites on each study plot with infiltrometers. Water flow after rainfall events were measured in 200 L collection tanks. A total of 212 rainfall events were monitored in the study. The maximum rainfall event was 101.1 mm and the maximum rainfall intensity was 68.58 mm/hr. The effect of prescribed burning on water flow was observed for two years after the treatments.

Changes in water flow were analyzed in terms of coefficients of regression between water flow after rainfall and the corresponding rainfall event. The most significant change in water flow occurred on plots experiencing the second fall burning. The observed increase in largely surface water flow relative to water flow from the control plots was attributed to reductions in infiltration rate after the prescribed fire treatments.

CONCLUSIONS

Studies in northwestern Chihuahua have resulted in valuable information on the importance of prescribed fire in protecting and, at times, enhancing forest resources. It is anticipated that the information obtained from these studies will be useful to the Secretariat of the Environment, Natural Resources and Fish in carrying out their the assigned responsibilities and operations.

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The Effect of Prescribed Burning to Control Brush Species on Buffelgrass Pastures in Sonora, Mexico[1]

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Abstract.—Vegetational changes were measured after 27 prescribed burns and 3 wildfires which occurred from 1982 to 1995 on buffelgrass pastures highly infested with brush in Sonora, Mexico. Densities of most undesirable brush species were reduced from 40 to 60% with fire. When prescribed burns occurred in good rainfall years the buffelgrass forage production exceeded (1.4 to 2.3 tons/Ha) that of unburned areas for three consecutive growing seasons after a single burn. Similar forage increases were also evident on pastures accidentally burned either once, twice or three times during alternated years. However, during dry growing conditions, less (1.2 tons/ha) buffelgrass forage was produced on burned than on unburned areas.

INTRODUCTION

Historically, fires have been reported to be rare in rangelands at the Sonoran desert, except in places where the desert integrates with the semidesert grassland (Humphrey 1974, Humphrey 1987) or in ranges where other grass species are sown (Cox et al. 1990). Fires occur on these areas, however, during falls and winters following years with exceptional rains which produce substantial grass and forb growth (Rogers and Steele 1980, McLaughlin and Bowers 1982, Cave and Patten 1984). Apparently, fire may have helped maintain an herb grassland community in parts of the Sonoran desert now dominated almost exclusively by shrubs, trees and cacti (Bock and Bock 1990).

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Buffelgrass (*Cenchrus ciliaris* L.) was introduced into Sonora, Mexico in the mid 1950's and by 1994 it was established in 400,000 hectares within the Sonoran desert (Martin et al. 1995). New pastures sown to buffelgrass produce 3-10 times more forage as compared to native rangelands (Ibarra et al. 1987), but buffelgrass yields usually diminish as less desirable brush species reinfest the pastures. Without maintenance practices, brush densities increase and pasture productivity may be seriously reduced within 10 years after planting (Hamilton and Scifres 1982).

Brush encroachment is a serious management problem on rangelands around the world and burning is a potential control technique in situations with adequate grass cover to support an intense fire (Trollope 1980, Cox et al. 1990, Wright 1990, Hodgkinson 1991). Winter burning of buffelgrass pastures in Texas has not had adverse effects on grass density and productivity (Mayeux and Hamilton 1983), similar results have been reported with summer burns in Sonora, Mexico (Ibarra et al. 1987).

This study was designed to:

1. Evaluate the effects of prescribed burning to reduce densities of brush species of low forage value for cattle in buffelgrass pastures,

2. To determine environmental variables at burning influencing brittlebush mortality with fire, and

3. To evaluate buffelgrass response to prescribed burning.

STUDY SITES

A series of experiments were conducted to evaluate the response of buffelgrass to brush control with fire in two regions typical of the Sonoran desert at Sonora, Mexico. A total of ten study sites with 3 replications each were selected. Eight study sites were on region 1 which is located on the central part of the state between Carbo and Hermosillo. Two more study sites were on region 2 which is located on the south-central part of the state between Mazatan and Tecoripa (Fig.1). Matorral arbosufrutescente

Figure 1. Location of regions of study. Region 1 typical of Matorral arbosufrutescente (Arbosufrutescent shrubland) vegetation type at central Sonora, and region 2 typical of Matorral arborescente (Arborescent shrubland) vegetation type at south-central Sonora, Mexico.

(arbosufrutescent shrubland) and matorral arborescente (arborescent shrubland) are the dominant vegetation types on regions 1 and 2, respectively. These vegetation types were selected because buffelgrass has been more extensively planted here (76% of the total planted area). Both vegetation types cover 5.5 million Ha of rangelands and represent about 30% of the total state land. (COTECOCA 1974).

Study 1

Conducted in region 1 on a 15 year old buffelgrass stand to reduce high densities of mesquite [*Prosopis juliflora* (Sw.) DC.], huisache [*Acacia farnesiana* (L.) Willd.], wait-a-minute (*Mimosa laxiflora* Benth.). Prescribed burning was applied on three plots 120.0 hectares each during June of 1985. Densities of mesquite, huisache and wait-a-minute averaged 327, 396 and 1,230 plants/ ha, respectively.

Study 2

Conducted on region 1 on a 4 year old buffelgrass pasture highly infested with brittlebush (*Encelia farinosa* A. Gray ex Torr.). Brittlebush density averaged 10,000 plants/ha. We took advantage of three accidental fires which occurred during June 1982, June 1984 and June of 1986. The first fire in 1982 covered 187 ha; the second fire in 1984 covered 90 ha, 55 ha of which were previously burned in 1982; and the third fire in 1986 covered 39 ha, 30 of which were burned in 1982 and again in 1984.

Study 3

Conducted at 6 study sites in region 1 on buffelgrass pastures infested with brittlebush. Brittlebush density varied among sites from 2,365 to 77,703 plants/ ha. Prescribed burns were applied once to eighteen 0.25 to 60.0 ha plots during June 1984, 1985 or 1986. Environmental variables at burning were measured and regressed against brittlebush mortality to determine the relationship between these variables and brittlebush mortality with fire.

Study 4

Conducted at 2 study sites on region 2 in buffelgrass pastures highly infested with acacia (*Acacia olygocantha* Gray), wait-a-minut, and white thorn

(*Acacia constricta* Benth.). Study sites were located at Tecoripa and Mazatan. Plant density per hectare between sites varied from 8,200 to 3,700; 4,580 to 2,771; and 2,490 to 270 for acacia, wait a minute and white thorn, respectively. At Tecoripa, prescribed burns were applied once to three 60.0 Ha plots during June 1986. Three additional 40.0 ha plots were burned at Mazatan during June 1995.

MATORRAL ARBOSUFRUTESCENTE

Occurs on bottom plains, arroyos, rocky hillsides, foothills, and small mountains. Principal textural classes are: sand, sandy loam, loamy sand, and loam. Precipitation, temperature and soil classification are present in Table 1. Dominant species are: grey thorn (*Condalia* spp. Cav), western redbud [*Caesalpinia pumila* (Britt. & Rose) Hermann], brittlebush, ironwood (*Olneya tesota* A. Gray), blue paloverde (*Cercidium floridum* Benth.), foothill paloverde [*Cercidium microphyllum* (Torr.) Rose & Johnston], mesquite, organpipe cactus [*Lemaireocereus thurberi* (Engelm.) Britt. & Rose], jumping choya (*Opuntia fulgida* Engelm.), spurge (*Euphorbia* spp. L.), rothrock grama (*Bouteloua rothrockii* Vasey), and six-weeks three-awns (*Aristida adscencionis* L.) (COTECOCA 1974). Other species present on the study sites include: desert hackberry (*Celtis pallida* Torr.), box thorn (*Lycium andersonii* A. Gray), horse tail (*Coniza* spp. L.), baccharis (*Baccharis sarothroides* A. Gray), huisache, wait-a-minute, croton (*Croton sonorae* Torr.), graythorn [*Condaliopsis lycioides* (A. Gray) Suesseng.], lycium (*Lycium berlandieri* Dunal), mascagnia [*Mascagnia macroptera* (Sessé & Moc.) Niedenzu.] and purgeroot [*Jatropha cardiophylla* (Torr.) Muell.].

MATORRAL ARBORESCENTE

Occurs on bottom plains, along arroyos, rocky hillsides, foothills, and small mountains. Principal textural classes are: Loamy sand, sandy loam, loam, sandy clay loam and clay loam. Precipitation temperature and soil classification are present in table 1. Dominant species are: coursetia (*Coursetia glandulosa* A. Gray), lysiloma [*Lysiloma divaricata* (Jacq.) J. F. Macbride], paloblanco [*Ipomoea arborescens* (Humb. & Bonpl.) G. Don], mesquite, organpipe cactus, chinese grama (*Cathestecum brevifolium* Vasey & Hack.), and rothrock grama (COTECOCA 1974). Other species present on the study sites include: wait-a-minut, mallow (*Abutilon californicum* Benth.), acacia, mesquite, bursage [*Ambrosia ambrosioides* (Cav.) Payne], white thorn, palo brasil (*Hematoxylon brasiletto* Karst), randia (*Randia thurberi* S. Wats.), vervain (*Lantana horrida* H. B. K.), croton, box thorn, and gray thorn.

METHODS AND MATERIALS

Field Measurements

Twenty seven plots (9 sampling sites) were intentionally burned once from June 1985 to June 1995 on

Table 1. Site characteristics and soil classification of two study regions.

Region[a]	Elevation m	Total precipitation mm	Mean annual temperature °C	Soil classification[b]
1	50-900	250-400	22-26	Yermosols, Xerosols, Lithosols, Luvisols, Kastanozems & Regosols
2	70-1,200	300-650	22-26	Kastanozems, Xerosols, Yermosols, Lithosols, Gleisols & Phaeozems

[a] *Region 1: Matorral Arbosufrutescente (Arbosufrutescent shrubland), Region 2: Matorral Arborescente (Arborescent shrubland) vegetation sub-types (COTECOCA 1974)*

[b] *Data from (FAO-UNESCO 1975)*

buffelgrass stands established from 1970 to 1990. Three additional plots (1 sampling site) which accidentally burned once, twice and three times, respectively, within the same period, were also included on the study. Selected plots had excellent to regular buffelgrass stands, but all show different intensities of brush invasion. Plot size varied from 0.25 to 60.0 hectares.

Firelines 6-meter wide were dozed around each plot. All plots were burned in June as headfires on relatively flat terrain following fire plan technology for non-volatile fuels as described by Wright (1973). Prescribed burns were conducted with drip torches between 9 and 10 A. M., while wildfires occurred in June from 1 to 3 P. M. Three plots were burned at each sampling site. Unburned checks were left at each study site for comparisons. Treated and untreated plots were fenced to protect from cattle grazing one year prior to burning and during three years after treatment application.

Immediately before burning, several measurements were taken to describe environmental conditions. Air temperature and relative humidity were measured with a sling psychrometer. Average wind speed 2 meters above ground was recorded with a totalizing anemometer (Clark et al. 1981). Fine fuel load was estimated by clipping, to within 5 centimeters of soil surface, 5 one square meter quadrats per plot. Fine fuel from each quadrat was separated into standing green, standing dead and ground litter components. Subsamples of each component were weighed in the field immediately after clipping, oven-dried at 50 °C to constant weight, then reweighed to determine fine fuel moisture content on a dry-weight basis for each component. Total fine fuel load at the time of burning was estimated by adding standing green, standing dead and ground litter components. Soil samples were taken from the surface 5 centimeters at 5 random locations. Samples were weighed, then dried at 105 °C for 48 hours and reweighed. Soil moisture content was determined gravimetrically.

Buffelgrass cover and height was estimated before and after burning on three 20 meter permanent transects per plot (Canfield 1941); and grass density on three 20 square meter quadrats per plot. Forage production was estimated at the end of each summer growing season by clipping 10 to 20 one square meter quadrats randomly placed in each plot. Forage samples were dried in a forced-draft oven at 40 °C for 72 hours and weighed. All yields are reported on a dry weight basis.

Brush density was estimated before and after prescribed burning on tree randomly selected 30 square meters quadrats per plot. Brush mortality was estimated after prescribed burning by comparing plant densities before and after treatment application. Brush mortality after the wildfires was estimated by comparing plant densities on burned against unburned adjacent areas.

DESIGN AND STATISTICAL ANALYSES

Experimental design was a completely randomized block with three replications, except the study site where accidental burns occurred. Data collected was subjected to analysis of variance at each study site. When F values were significant ($P \geq 0.05$) means were compared using Duncan's Multiple Range Test (Steel and Torrie 1980). Simple linear regression and correlation analysis were used to determine the relationship between several variables and brittlebush mortality with fire. Because of incomplete data at some study sites, only information from 18 burned plots was included in these analysis. Brittlebush mortality (dependent variable) was regressed against independent variables listed in Table 2.

Table 2. Independent variables considered in analyzing brittlebush mortality in buffelgrass pastures of central Sonora, Mexico.

Variable	Minimum	Maximum
Fine fuel load, standing dead (Tons/Ha)	0.15	3.43
Fine fuel load, standing green (Tons/Ha)	0.03	0.10
Fine fuel load, ground litter (Tons/Ha)	0.01	0.89
Fine fuel load, total (Tons/Ha)	0.19	4.42
Moisture content, standing dead (%)	1.7	4.9
Moisture content, standing green (%)	48.3	78.6
Moisture content, ground litter (%)	2.1	7.8
Soil moisture (%)	0.3	1.8
Buffelgrass density (Plants/m²)	3.2	9.5
Buffelgrass basal cover (%)	5.4	18.3
Brittlebush density (Plants/m²)	2,365.0	77,703.0
Brittlebush canopy cover (%)	18.6	79.6
Air temperature (_C)	26.3	37.5
Relative humidity (%)	12.5	26.7
Wind speed (Km/h)	3.8	15.9

RESULTS AND DISCUSSION

Study 1

Brush and buffelgrass density were adequate to impair fine fuel production and continuity. Fine fuel load at the time of burning averaged 2.3 tons D.M./ ha, 37 % of which was on the ground as litter. Wind speed averaged 5 Km/h, relative humidity was 20% and air temperature 21 °C. Fine fuel water content was 3.5%. The burns were uniform and blackened an estimated 90% of the surface area. About 10% of the area within plots was not burned because of the lack of fuel continuity.

Total precipitation was 76 and 300 mm in 1985 and 1986, respectively. The sprout of most brush species after burning was limited due to low rainfall which occurred during the summer of 1985. As a result, plant mortality estimations were made after the 1986 summer growing season. Although all brush species were damaged by the fire, susceptibility of plants to fire was variable among species (Table 3). Similar results have been reported on the upper Sonoran Desert region (Cave and Patten 1984, Patten and Cave 1984).

Species more susceptible to fire were desert hackberry, box thorn, horse tail and baccharis which were reduced from 73 to 100%. Densities of huisache, brittlebush, wait-a-minut, organpipe, foothill paloverde, mesquite and croton were reduced by 31 to 60%. Plants less susceptible to fire were graythorn, iron wood, lycium, mascagnia and purgeroot which were reduced by less than 22%.

Because most brush species are more damaged by fire on dry years as compared to wet years (Wright and Bailey 1982, Smith and Tainton 1985) the susceptibility of some species to fire on this study was possibly magnified by the dry conditions prevailing during the year of burning. Foothill paloverde and ironwood are plants of high forage value for cattle, but both are highly susceptible to fire. Consequently, the use of fire as a tool for controlling undesirable species on buffelgrass pastures has limited future on areas where brush species of high forage value are present.

Green leaves on buffelgrass plants appeared in less than 10 days after burning, but lack and delayed

Table 3. Brush mortality obtained two growing seasons after the application of a hot summer fire in June 1985 on buffelgrass pastures highly infested with brush at central Sonora, Mexico.

Scientific name	Common name	Mortality percent	Livestock preference[a]
Celtis pallida	Desert hackberry	100	3
Lycium andersonii	Box thorn	100	3
Coniza spp.	Horse tail	100	3
Baccharis sarothroides	Baccharis	73	3
Acacia farnesiana	Huisache	60	2
Encelia farinosa	Brittlebush	55	2
Mimosa laxiflora	Wait-a-minut	51	2
Lemaireocereus thurberi	Organpipe	46	3
Cercidium microphyllum	Foothill Paloverde	42	1
Prosopis juliflora	Mesquite	38	2
Croton sonorae	Croton	31	2
Condaliopsis lycioides	Graythorn	22	3
Olneya tesota	Iron wood	21	1
Lycium berlandieri	Lycium	16	3
Mascagnia macroptera	Mascagnia	5	3
Jatropha cardiophylla	Purgeroot	3	3

[a] *Animal preference obtained from botanical composition of diet trials conducted at CIPES. Species preference are variable among years, regions, breeds and grazing seasons and intensities of use. Preference ratings: (1) high, (2) light to moderate, (3) none.*

rainfall limited plant growth on summer 1985. Buffelgrass density and cover were similar on burned and unburned plots during 1985, but grass height and production were significantly (P≥0.05) greater on unburned checks (Table 4). Buffelgrass plants were 15 cm taller and forage production per hectare was 1.2 tons greater on untreated checks as compared to burned plots. Similar results are reported by Hamilton and Scifres (1982), after burning buffelgrass pastures in Texas.

Significant growth on burned buffelgrass plants did not occur until summer of 1986, when three storms occurred during July and August. Weekly amounts exceed 25 mm and by the end of the summer, buffelgrass forage production on burned areas (4.0 tons/ha) was 53% greater than unburned checks (Table 5). Buffelgrass density, cover and height on burned plots was also 30 to 69% greater (P≥0.05) as compared to untreated checks.

Study 2

Precipitation at the study site was above normal at all years, except at 1985 where rainfall was 13% below the 20 years average. Total precipitation was 349, 529, 477, 285, 443 and 370 mm in 1982, 1983, 1984, 1985, 1986 and 1987, respectively. Before the 1982 fire, the buffelgrass stand was on regular condition but highly infested with brittlebush. Brittlebush densities averaged 10,000 plants/ha. Buffelgrass standing crop and litter averaged 4.1, 3.5, and 3.2 tons/ha on areas which escaped from fire on 1982, 1984 and 1986, respectively. Because the three wildfires occurred within the same pasture and all were manually controlled, we assume similar vegetational characteristics on burned and unburned sites.

Accidental fires on buffelgrass stands significantly reduced densities of both, old brittlebush plants (> 10 cm) and seedlings (< 10 cm) (Fig. 2). The greatest decline on the density of mature plants occurred after the 1982 and 1984 fires. Old brittlebush plants averaged 6,640/ha on adjacent unburned areas; 4,515 plants/ha on areas which burned once during 1882; 1,355 plants/ha on areas that burned twice during 1882 and 1984; and 54 plants/ha on areas that burned three times during 1982, 1984 and 1986. Brittlebush seedlings were more susceptible than old plants and densities drastically declined after the 1982 fire. Brittlebush seedlings averaged 3,154 plants/ha on adjacent unburned areas; 1,263 plants/ha on areas that burned once during 1982; 51 plants/ha on areas that burned twice during 1982 and 1984; and 26 plants/ha on areas that burned three times during 1982, 1984 and 1986.

Buffelgrass forage production on sites which accidentally burned either once, twice or three times was (3.18 to 4.62 tons/ha) 30 to 45% greater as compared to unburned adjacent areas (Table 6). Similar forage changes are reported after burning of buffelgrass pastures for brush control in Texas (Mayeux and

Table 4. Buffelgrass density, cover, height and production one summer growing season after the application of prescribed burning in June 1985 at central Sonora, Mexico.

Parameter	Treatments: Burned	Treatments: Checks	Difference	Percent Increase
Density (plants/m²)	8.4 a	7.5 a	+ 0.9	12
Cover (%)	5.8 a	5.4 a	+ 0.4	7
Height (cm)	24.6 b	39.5 a	- 14.9	38
Production (tons.D.M/ha)	1.4 b	2.6 a	- 1.2	46

Means in rows followed by the same letters are not significantly different (a=0.05) by Duncan's multiple range test.

Table 5. Buffelgrass density, cover, height and production two summer growing seasons after the application of prescribed burning in June 1985 at central Sonora, Mexico.

Parameter	Treatments: Burned	Treatments: Checks	Difference	Percent increase
Density (plants/m²)	10.0 a	6.8 b	+ 3.2	47
Cover (%)	8.8 a	5.2 b	+ 3.6	69
Height (cm)	74.0 a	57.0 b	+ 17.0	30
Production (tons.D.M/ha)	4.0 a	2.6 b	+ 1.37	53

Means in rows followed by the same letters are not significantly different (a=0.05) by Duncan's multiple range test.

Figure 2. Changes in brittlebush densities one growing season after the occurrence of one, two or three alternated accidental summer fires during June 1982, 1984 and 1986 on a buffelgrass community highly infested with brittlebush at central Sonora, Mexico.

Hamilton 1983). Excellent precipitation among years on this study, may account for the consistently greater forage production obtained on sites which burned more than once. Hamilton and Scifres (1982) report that a second burn may reduce buffelgrass productivity if moisture is limited for plant growth.

Table 6. Buffelgrass total forage production (Tons D.M./Ha) after two summer growing seasons on unburned sites and after the occurrence of one, two, or three alternated accidental summer fires during 1982, 1984 and 1986 on buffelgrass pastures highly infested with brittlebush at central Sonora, Mexico.

	Treatments			Percent
Parameter	Burned	Checks	Difference	increase
After 1982 burn only	4.62	3.18	+ 1.44	45
After 1982 & 84 burns	3.18	2.45	+ 0.73	30
After 1982, 84 & 86 burns	4.37	3.06	+ 1.31	43

Means in rows followed by the same letters are not significantly different (α=0.05) by Duncan's multiple range test.

Study 3

Brittlebush mortality (dependent variable) was compared to independent variables listed in Table 2. By using simple linear regression analysis, it was found that the single variable most significantly ($P \geq 0.05$; $r^2 = 0.93$) related to brittlebuh fire injury was total fine fuel load at the time of burning (Fig. 3). Although brittlebush mortality with fire varied from 8 to 100% among the 18 study sites, brush mortality linearly increased as total fine fuel load increased. By using regression equation, 29% brittlebush mortality can be expected when burning buffelgrass pastures with 1.0 ton/ha of total fine fuel load; and close to 100% mortality with above 4.5 tons/ha of total fine fuel at the time of burning.

Limited fuel amounts at burning resulted in low brush mortality rates. Most brittlebush plants in buffelgrass pastures which were burned with less than 0.7 tons/ha of fine fuel escaped from the fire. Injured plants which were partially burned regrowth vigorously one growing season after the fire. This may explain why low fuel load burns may enhance rangeland productivity in the Upper Sonoran Desert (Cave and Patten 1984).

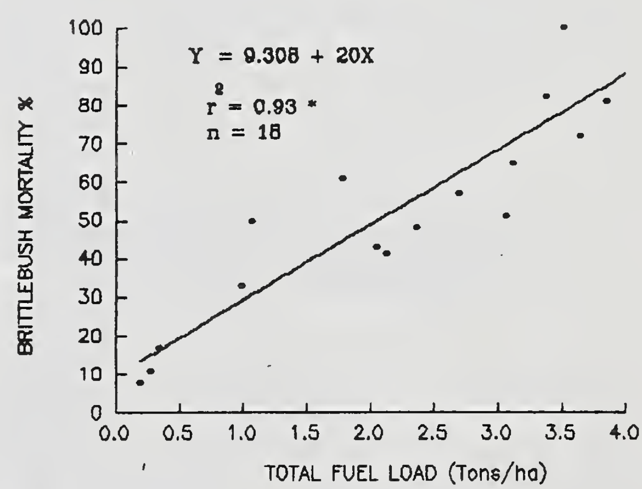

Figure 3. The effect of fine fuel load (Tons/ha) at the time of burning on brittlebush mortality (%) after the application of 16 prescribed burning trails on buffelgrass pastures at central Sonora Mexico.

Study 4

Precipitation at Tecoripa site was close to normal during the study period. Total precipitation was 515, 463 and 490 mm in 1986, 1987 and 1988, respectively. Total fine fuel load at the time of burning averaged 6.19 tons/ha, 73% of which was standing biomas and the remaining 27% as ground litter. Wind speed averaged 12 Km/hour, relative humidity was 26% and air temperature 28 °C. Average fine fuel water content was 12%. The burn was fast and uniform and blackened more than 95% of the surface area because the fuel load was high.

Although all brush species were damaged by the burn, susceptibility of plants to fire was variable among species (Table 7). Species more susceptible to fire were wait-a-minute, mallow, acacia, and mesquite which were reduced from 63 to 79%. Brush species less susceptible to fire were bursage, white thorn, palo brasil and randia which were reduced from 47 to 53%.

Buffelgrass forage production was significantly greater ($P \geq 0.05$) during 1986, 1987 and 1988 on burned plots as compared to unburned checks (Table 8). Total standing crop on burned areas varied from (6.82 to 8.15 tons/ha) among years and was 17 to 33% greater as compared to untreated checks. During a three year period, buffelgrass burned plots produced 5.6 tons/ha of additional forage as compared to untreated plots.

Buffelgrass basal cover and density were not significantly ($P \geq 0.05$) affected by fire at any year of evaluation (Table 8). Buffelgrass cover varied among years from 15.1 to 18.5% on burned plots, and from 13.9 to 14.8% on untreated checks. Plant density varied among years from 8.6 to 10.6 plants/m² on burned plots, and from 8.2 to 9.5 plants/m² on untreated checks.

Results obtained at Mazatan study site one summer growing season after burning were very similar compared to these of Tecoripa. Precipitation at Mazatan during 1995 was 525 mm. Total fuel load at the time of burning was 4.3 tons/ha, 84% of which

Table 7. Brush mortality obtained three growing seasons after the application of prescribed burning in June 1986 on buffelgrass pastures highly infested with brush at Tecoripa Sonora, Mexico.

Scientific name	Common name	Mortality percent
Mimosa laxiflora	Wait-a-minut	79
Abutilon californicum	Mallow	78
Acacia olygocantha	Acacia	76
Prosopis juliflora	Mesquite	63
Ambrosia ambrosioides	Bursage	53
Acacia constricta	White thorn	52
Haematoxylon brasiletto	Palo brasil	52
Randia thurberi	Randia	47

Table 8. Buffelgrass forage production (Tons/Ha), basal cover (%), and density (Plants/m²) 1, 2 and 3 summer growing seasons after the application of prescribed burning to reduce brush density on buffelgrass pastures at Tecoripa, Sonora, Mexico.

Year	Treatments		Percent	
	Burned	Checks	Difference	Increase
Forage Production				
1986	6.82 a	4.57 b	+ 2.25	33
1987	7.75 a	5.82 b	+ 1.93	25
1988	8.15 a	6.73 b	+ 1.42	17
Basal Cover				
1986	15.6 a	14.2 a	+ 1.4	9
1987	18.5 a	14.8 a	+ 3.7	20
1988	15.1 a	13.9 a	+ 1.2	8
Density				
1986	10.6 a	8.2 a	+ 2.4	23
1987	10.3 a	9.1 a	+ 1.2	12
1988	8.6 a	9.5 a	+ 0.9	10

Means in rows followed by the same letters are not significantly different ($\alpha=0.05$) by Duncan's multiple range test.

Table 9. Brush mortality (%) obtained one growing season after the application of prescribed burning in June 1995 at buffelgrass pastures highly infested with brush at Mazatan, Sonora, Mexico.

Scientific name	Common name	Mortality percent
Abutilon californicum	Mallow	81
Mimosa laxiflora	Wait-a-minut	76
Haematoxylon brasiletto	Palo brasil	66
Lantana horrida	Vervain	62
Croton sonorae	Croton	60
Lycium andersonii	Box thorn	58
Acacia olygocantha	Acacia	58
Ambrosia ambrosioides	Bursage	58
Randia thurberi	Randia	49
Condaliopsis lycioides	Graythorn	47

was present as standing fuel and the remaining 16% was present as ground litter. High fuel load allowed a fast homogeneous burn that blackened more than 90% of the area burned.

Brush species more susceptible to fire were mallow and wait-a-minute which declined by 81 and 76%, respectively (Table 9). Densities of palo brasil, vervain, croton, box thorn, acacia and bursage were reduced from 58 to 66%. Brush species less damaged by fire were randia and graythorn which were controlled by 49 and 47%, respectively.

MANAGEMENT IMPLICATIONS

Brush infestations are common on areas that have been mechanically cleared of brush and seeded to buffelgrass. Increased cost of chemical and mechanical treatments have contributed to the growing interest in fire as an alternative method for the restoration of buffelgrass pastures infested with undesirable brush species.

Prescribed burning is not an effective tool for the control of undesirable brush species in deteriorated or overgrazed buffelgrass pastures; unless reclamation techniques are used to increase forage fuel amount and continuity for adequate burning. Susceptibility of plants to fire is variable among species. Although at least 2.0 tons/ha of total fine fuel is required at the time of ignition to damage most plants, some brush species may require up to 4 tons/ha. Densities of most undesirable plants were reduced from 40 to 60% with prescribed burning. However, because fire is not a brush selective control practice, its use for controlling undesirable species has limited future on buffelgrass stands where undesirable and desirable plants are mixed.

Buffelgrass is a fire resistant plant capable of fast recovery if adequate moisture prevails after burning. When burned on good rainfall years the plant responded by a flush of summer growth during the year of burning and cumulative forage production exceeded (1.4 to 2.3 tons/ha) that of unburned areas for three consecutive growing seasons after a single burn. Similar forage increases were also evident on pastures accidentally burned either once, twice or three times during alternated years. However, during dry growing conditions, less (1.2 tons/ha) buffelgrass forage was produced on burned than on unburned areas. Prescribed burning had no adverse effect on buffelgrass under the Sonoran desert conditions. Buffelgrass density and cover tend to increase (9 to 69%) in burned pastures when fire was followed by average or above average precipitation. Substantial yield increases are expected after high precipitation occurs following burning. However, because rainfall during active plant growth is below normal in 3 or 4 of 10 years, the potential use of prescribed burning is greater on the moister parts of the Sonoran Desert.

Fire plays a very important role in buffelgrass pastures to control density and cover of undesirable brush species. Prescribed fires can effectively reduce densities of hackberry, box thorn, horse tail, brittlebush, wait-a-minute, mesquite, huisache, croton, mallow, acacia, palo brasil, vervain, white thorn and bursage. Other benefits of burning buffelgrass pastures include: increase of access and visibility for grazing animals, increase of cattle distribution and forage use, reduction of spittlebug densities and pasture damage, increase of grass density and cover, increase of forage quantity and quality, and reduction of coarse grass material at the plant bases.

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Clean Air and Healthy Ecosystems: Managing Emissions from Fires

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Abstract.—After nearly a century of avid fire suppression, land managers are substantially increasing prescribed burning to meet ecosystem management objectives. As scientists and managers we need to accurately quantify the capacity of airsheds to assimilate smoke and related atmospheric pollutants from wildfire and prescribed fire within acceptable limits for air quality. Conversely, we need to quantify increases in ecosystem health that result from prescribed fire, as well as the ecological cost of fire suppression. Resolutions for prescribed burning programs to protect both soil, water and air quality and foster healthy ecosystems are presented. This includes a discussion of revised models and current efforts to quantify how prescribed fire can be used to offset wildfire emissions.

INTRODUCTION

Air resource management and ecosystem management are inseparable because air, in addition to water and soil, is a fundamental physical component of natural ecosystems. Many biotic components of ecosystems depend on suitable soil, water and air quality for their survival. In general, wildfires have a greater capacity for adverse effects on natural resources compared to prescribed fire. Thus, conducting prescribed fires that protect both soil, water and air quality and overall ecosystem health is warranted.

Reasons for Prescribed Fire in Forests and Grasslands

There are a variety of reasons for using prescribed fire in forests and grasslands. The incidence of costly, catastrophic wildfire is reduced as the amount of flammable organic matter on the forest floor declines (Martin et al. 1989). It can provide open habitat and food for wildlife and maintain vegetation species composition in fire-adapted ecosystems. Prescribed fire is also used to dispose of debris from timber harvest, to manage insects and disease, to enhance scenic values, and to protect people and property at the urban ecotone from wildfire hazards (NWCG 1989). This paper primarily concerns prescribed fire for the purpose of ecosystem management. Thus, it pertains to ecosystems where fire is a significant natural component and the natural fire regime has been interrupted through human intervention.

Reasons to Protect Ecosystems from Unacceptable Fire Effects

There are many reasons to protect ecosystems from unacceptable effects of fire emissions. The foremost reason is to prevent health hazards to firefighters and the general public. Another is to maintain scenic values and visibility for both aesthetic and safety reasons, and to respond to public demand for clean air. Catastrophic wildfires can pollute surface and ground water, while such detrimental effects are less likely with more moderate fires. Soil quality can be impaired as a direct result of the temperature and duration of heating, as discussed below. Finally, we must meet legal mandates for limits to anthropogenic emissions.

Water Quality

The extent of detrimental effects and length of recovery time for aquatic ecosystems is a function of

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fire intensity and size of the area burned, intensity of subsequent storms and the occurrence of other disturbances. In an experiment comparing the results of moderate, severe, and no fire on watersheds, annual nitrate loss from severely burned areas was forty times greater than unburned areas and seven times greater than in moderately burned areas (Riggan et al. 1994). Volume-weighted nitrate concentrations in streams were 1.7 times higher for severe burns than for moderate burns. Moderate burns resulted in nitrate concentrations three times greater than in unburned watersheds. Ammonium ion flux was also affected. Sedimentation associated with a moderate burn was about 40 percent that of a severe burn, while sedimentation was negligible in unburned watersheds. Severe burns resulted in Federal water quality standards violations for nitrate and may also have contributed to pollution of the aquifer.

Fire effects on aquatic ecosystem are generally greatest in headwater streams which can shift from a heterotrophic system that is light-limited to an autotrophic system (Minshall et al. 1989). Increases in primary production are associated with increases in light and nutrients, especially in oligotrophic (low-nutrient) waters. Although nitrogen concentrations may increase in the waterbodies, phosphorus can limit primary and secondary production despite increases in solar radiation. Similarly, there is little adverse effect on water quality within streams larger than forth order, even with wildfire (Minshall et al. 1989).

The range of effects on water quality and stream biota was postulated following the Yellowstone National Park wildfires that occurred in 1988 (Minshall et al. 1989). Few adverse effects on water chemistry were predicted other than sediment suspension, but there were anticipated increases in allochthonous carbon and changes in nutrient cycling. Aquatic biota can be adversely affected by high water temperatures, changes in water chemistry, and an acute change in food supply associated with fire. Changes in algal dominance were also predicted. Water temperature can increase due to loss of protective vegetative cover and then stress coldwater fishes. An increase in the suspended sediment load can be associated with increases in streamflow. As a result, fine sediment accumulation can reduce populations of benthic macroinvertebrates that utilize a more coarse substrate. Subsequent biotic recovery may require three or more years. Such changes in aquatic ecosystems would be analogous to terrestrial ecosystem succession that occurs after fire.

Soil Quality

In general, wildfires have a greater capacity for adverse effects on soils compared to prescribed fire. Prescribed fire generally burns at a cooler temperature and is less detrimental to soil quality than wildfire. It produces lower soil temperature, maintains soil structure and microbes, and results in less nutrient loss and erosion (DeBano 1989). Also, large trees and protective organic matter on the ground are generally less disturbed by prescribed fire than wildfire or Prescribed Natural Fire (PNF).

A prescribed fire experiment in pine forest of South Carolina showed limited effects on soils, nutrient cycling and water quality, including only minor effects on soil chemistry (Richter and Ralson 1982). Fire can alter the physical, chemical and biological properties of soil. It can reduce soil organic matter more rapidly than natural decomposition processes, especially in arid ecosystems. Some effects of fire on soil quality are a function of the resultant soil temperature and duration of heating. For example, only severe burns alter particle size and soil structure. The amount of nutrients loss to volatilization is a function of the fire temperature and the soil moisture gradient. Severe fires can also cause surface soils to be hydrophobic, and therefore impact infiltration. Porosity may decline as pores become clogged with fine particles. Infiltration may also be reduced due to litter losses. The subsequent decline in terrestrial uptake results in nutrient mineralization and leaching. Hot, explosive fires sterilize the soil (O'Hanlon 1995) resulting in a loss of microflora, e.g., bacteria that are key to the nitrogen cycle and mycorrhizal fungi. Thus, the use of cooler, less intense prescribed fire can reduce the potential for adverse effects of wildfire on soil quality.

Air Quality

One approach to achieving an ecologically appropriate level of fire needed for both restoring and maintaining healthy forest ecosystems while maintaining acceptable air quality is to practice smoke management. Smoke management has two primary components, i.e., management of the fire process itself to minimize the generation of pollution and

utilizing the dispersive nature of the atmosphere to dilute emissions before impacting people and population centers. The former, managing the fire itself, is much preferred in pollution prevention. However, fire process management to minimize smoke production is often at odds with other aspects of fire management objectives.

Fire Management to Minimize Smoke Production

Fire in ecosystems is either planned or unplanned. Unplanned fire generates significant levels of pollution emissions, causes landowners and managers loss of commodity and amenity values and often threatens the public well being. Thus, unplanned fires have a number of negative consequences. However, these negative consequences are often weighed against positive ecological benefits in certain fire-adapted ecosystems. As a result, managers recognizing both the ecological values and the social costs, have developed fire management plans or prescriptions. The ignition can be either unplanned or planned, but the fire is monitored on a regular basis to ensure that it meets the prescription.

Fire prescriptions identify desired fire intensities in order to achieve ecological objectives. These are ecosystem dependent. From a smoke production standpoint, there are a few simple concepts to consider. What sort of fire will minimize the generation of smoke? What temperature fire will minimize the release of particles in the 10 micron and smaller size range? What type of fire will generate enough heat intensity to loft the smoke well above the forest canopy so that it doesn't become trapped in the area of the burn or entrained in a specific drainage flow carrying it into a nearby urban area?

Utilizing Atmospheric Dispersion

The turbulent character of the atmosphere has been recognized for most of history as providing an option for removing air pollution from where it is generated. As soon as people started living in shelters they developed the capability to remove smoke from the fires they used for cooking and heating. In modern times, this has been made into a science with smokestacks designed to loft pollutants well above the land surface in hopes that they will disperse to below detectable levels by the time they reach the ground.

Forest burning has traditionally generated smoke that caused hazy conditions in the regions where the burning has occurred. One aspect of smoke management includes attempting to generate enough energy in the fire so that the smoke clears out of the forest canopy and forms a plume that is well above the ground. This plume then is free to disperse in the atmosphere. A somewhat more sophisticated aspect of smoke management is to try to limit burning to times and locations such that wind will transport the smoke away from population centers. A third variation is to allow the fire to burn only so long as its smoke is not approaching a population center or other smoke-sensitive area.

All of these techniques require some intelligence about wind speed, direction, and spatial distribution. This includes not only current conditions but what conditions will be when the burn is scheduled to begin and how conditions may change during the duration of the burn. Also it is necessary to know something about the dispersive capacity of the atmosphere, which is referred to as atmospheric stability. Similarly, it is necessary to analyze the current and anticipated future atmospheric stability. Some of this needed information comes from measurements, but measurements are often inadequate and unable to predict into the future. Thus, models of atmospheric conditions and of the potential responses of smoke to these atmospheric conditions are needed.

PROBLEM STATEMENT

Federal forest and grassland fire policies recognize fire as a critical natural process. At a landscape level fire is used "to protect, maintain, and enhance resources and, as nearly as possible, be allowed to function in its natural ecological role" (USDI and USDA 1995). As scientists and managers we need to quantify the capacity of (specific) airsheds to assimilate smoke and related pollutants from wildfire and prescribed fire within acceptable limits for soil, water and air quality. Conversely, we need to quantify increases in ecosystem health that results from prescribed fire, as well as the ecological cost of fire suppression. Both regulators and land managers need accurate emissions estimates to quantify effect of prescribed fire on air quality. Because human health depends on ecosystem health, it is reasonable to assure that prescribed burning programs provide both clean air and healthy ecosystems.

SYNTHESIS AND CONCLUSIONS

Issues Related to Ecosystem Management

In 1995 it was reported that only about one quarter of one percent of land administered by the Forest Service in the western United States was being treated with prescribed fire (Sampson 1995). However, many National Forests are substantially increasing the amount of controlled burning, or adjusting the seasonal timing when burning is conducted. Yet, fire management and atmospheric pollutants both pose significant threats to wilderness ecosystems and cause a departure from natural conditions (Cole andLandres 1996). For example, fire suppression provides a competitive advantage to shade-tolerant species that does not occur under natural fire regimes. Nearly all wilderness areas have felt the impact of regional atmospheric pollutants which often travel tremendous distances (National Academy of Sciences 1993). While much progress has been made in understanding complex ecosystems, we need to further understand the spatial and temporal variations of natural fires and the complexity of factors that determine these landscape patterns (Cole and Landres 1996).

Issues Related to Air Resource Management

Two important issues related to air resource management are the need to minimize pollutant emissions and to meet legal requirements for air quality. The effect of Forest Service activities on air quality must be adequately addressed before implementing prescribed fire for ecosystem management (USDA Forest Service 1994). We need to disclose environmental effects from prescribed fires and determine whether such actions conform to State Implementation Plans for meeting and maintaining the national ambient air quality standards, including existing PM-10 standards.

Minimize Pollutant Emissions

Fires affect atmospheric chemistry. They emit carbon monoxide, carbon dioxide, nitrogen oxides, hydrocarbons such as methyl bromide and benzene. Nitrogen oxides (NO_x) and carbon monoxide can react to produce tropospheric ozone pollution and about 38 percent of tropospheric ozone originates from biomass burning (Pease 1992). However, NO_x is only produced from biomass burning at high temperature, such as in logging slash piles or high-intensity wildfire (NWCG 1989). More than 90 percent of particulates from prescribed fire are less than 10 microns in diameter, which can impair visibility (USEPA 1992). Fires also emit toxic substances including acetaldehyde, acrolein, formaldehyde, and toluene (USEPA 1992).

Concerns for fire's contribution to global changes have been expressed. Methyl bromide (CH_3Br) contributes to the destruction of the stratospheric ozone layer (Cicerone 1994). Carbon dioxide emissions from fire contribute to concentrations of greenhouse gases; however, prescribed fires are low-intensity relative to wildfire and therefore emit less carbon dioxide. Some studies also estimate that the carbon dioxide is ultimately absorbed by new vegetative growth following prescribed fire, resulting in no net increase of greenhouse gases (O'Hanlon 1995, USDI and USDA 1995). Larger contributors could include global grassland burning, especially in African savannas, and rainforest destruction (O'Hanlon 1995).

Legal Requirements

Federal land managers must meet all applicable regulations and standards, including the National Ambient Air Quality Standards (NAAQS) for particulates (PM-10), carbon monoxide, ozone, and nitrogen dioxide. State or county open-burning regulations address prescribed fire emissions. In addition we formulate cooperative, interagency agreements to help achieve our mutual goals and participate in the development and revision of State Implementation Plans (SIPs) to meet standards in non-attainment areas and maintain state and federal standards in attainment areas. Potential environmental impacts of prescribed fire and corresponding mitigation measures are also identified during the National Environmental Policy Act (NEPA) scoping process.

Four areas of the Clean Air Act refer to prescribed fire:

1. NAAQS and control measures to meet those standards apply to areas that are not in attainment;
2. Conformity to SIPs for air quality is required;
3. Fire is defined as a "movable, stationary source" that would apply under some permit enforcement provisions; and

4. Reasonable progress toward the National Visibility Goal of no anthropogenic impairment of visibility in Class I wilderness areas could include fire emissions.

The American Lung Association won a legal case requiring EPA to regulate smaller particle sizes than the current regulations, due to health effects. Therefore, PM-2.5 standards (particulate matter less than 2.5 microns in diameter) are being formulated.

Section 190 of the Clean Air Act as amended (1990) requires prescribed burns in areas that do not meet national standards to have acceptable control measures to reduce emissions. There are two classes of control measures. Reasonable Available Control Measures (RACM) apply to moderate non-attainment areas and Best Available Control Measures (BACM) apply to serious non-attainment areas. General RACM guidelines were published by EPA in 1992. BACM guidelines are contained in three subsequent EPA technical documents. More specific control measures are often included in SIPs or related publications (see ADEQ 1991). Lists of areas that do not meet national standards are published in the Federal Register.

Several groups are currently addressing legal issues related to prescribed fire and wildfire emissions. The Western States Air Resource (WESTAR) Council is developing a wildfire/prescribed fire project work plan with objectives, tasks, timelines and provisions for interagency coordination. Their Forest Health Initiative to Restore Ecosystems project will assess the impact of wildfire on public health and assist in ranking the priority of prescribed burning for ecosystem management. Best Available Control Measures are recommended for prescribed fires conducted to restore ecosystem health. WESTAR is also developing a regional air quality impact assessment of alternatives to prescribed burning. The Grand Canyon Visibility Transport Commission established by the 1990 Clean Air Act has a fire emissions inventory project to quantify emissions from prescribed fire and wildfire in 11 western states. EPA is developing a regional haze plan that could address prescribed fire with January 1997 as target date for the final rule. EPA has also considered conformity requirements for certifying that air quality goals are achieved in attainment areas.

Achieving Clean Air *and* Healthy Ecosystems

A combination of continuing research, judicious agency policies, monitoring and modeling, and interagency cooperation is key to the success of prescribed burning for ecosystem management purposes. The Forest Service (1994) developed a National Air Resource Management Strategy that lists nine strategies for air pollution from Forest Service activities, including fire management. These address training, technical and administrative tools, emission inventories, cumulative effects analysis, setting priorities, policy development, research, and developing national direction. For example, it is essential to work with states and other regulatory agencies in developing or revising SIPs, and to disseminate public information prior to and during the burn.

Best management practices for smoke are, to the extent practicable, practiced by the Forest Service. Smoke management is an integral part of burning plans. Guidelines for emission control measures are contained in USDA Forest Service documents (1992), EPA publications (1992), state guidance documents (Arizona Department of Environmental Quality 1991), and collaborative publications (NWCG 1985, 1989). Minimum objectives include "identify and avoid smoke-sensitive areas; reduce emissions; and disperse and dilute smoke before it reaches smoke-sensitive areas" (National Wildfire Coordinating Group 1985). This includes emergency plans to extinguish a fire that has become unacceptable due to air resource impacts. It is important for land managers to identify these "red flag" conditions and take immediate, appropriate action, i.e., not to rely on notification of unacceptable conditions from state air quality regulators.

An interagency MOU for smoke management in NM contains four objectives:

1. "to minimize the generation or impacts of smoke in New Mexico when prescribed burning is necessary for the conduct of accepted ecosystem management practices; alternative treatments will be encouraged and used where environmentally acceptable, technologically feasible, and economically reasonable;"

2. "to assure that no ambient air quality standards or air quality control regulations are violated;"

3. "to minimize visibility impacts from smoke in smoke-sensitive areas and in important views in Class I areas, especially during times of significant visitor use; and"

4. "to develop and implement a system to inventory emissions from prescribed fires and wildfires."

The MOU clarifies terminology, permit authority, procedures and conditions, and discusses any requirements for suppression, monitoring and reporting. It also initiates the development of more specific zone plans that identify smoke-sensitive areas, best management practices, training and monitoring needs.

Wildfire, Prescribed Fire (Planned Ignitions) and Prescribed Natural Fire (PNF)

There are many differences between wildfire, prescribed fire, and lightning ignitions in wilderness that are managed as prescribed natural fire. The amount of emissions over time and space, risk of smoke intrusion, and environmental consequences are important variables. Banta et al. (1992) compared smoke plume behavior from a prescribed fire and a wildfire using Doppler radar and lidar. Smoke from the prescribed fire tended to remain within a mixed atmospheric layer. In contrast, wildfire evoked a bent convection column of smoke that responded to changing meteorological conditions by bending.

PNF tends to burn for much longer periods of time than other prescribed fire, so there is a greater chance of encountering adverse weather conditions and subsequent smoke intrusions into population centers or scenic vistas. WESTAR recognizes wildfire smoke as most injurious to public health. This contrast is due to better smoke dispersion, mitigation measures, and less biomass combustion with prescribed fire.

Wildfire has a greater capacity to adversely impact soil, water and air resources. Conversely, the strategic use of prescribed fire can have multiple benefits on ecosystem health. A prescribed fire experiment located in a South Carolina pine forest showed limited effects on soils, nutrient cycling and water quality (Richter and Ralson 1982). Atmospheric effects included nitrogen and sulphur released from the forest floor. Lowest nitrogen concentrations occur in the surface litter, so prescribed fire may be less detrimental if subsurface soils are impacted less. Compared to control watersheds there were no significant changes in stream concentration for twelve anions due to prescribed fire treatments. There were also no effects on groundwater quality in the pine forests.

Meteorological conditions that favor smoke dispersion and restrict smoldering need to be projected and communicated with enough lead-time to respond to short-term changes in wind speed and direction, air mass stability and associated vertical mixing height, relative humidity or air temperature. Evening burns are more risky with respect to smoke impacts due to the possibility of inversions (NWCG 1989). Fires that burn for extended time periods are also more likely to encounter weather that is unsuitable for dispersion. For example, a 1994 wildfire in Idaho resulted in federal air pollution standards violations at a distance of 225 km near Missoula, Montana (Sampson 1995). Conversely, prescribed fire has less potential to contribute to nonattainment of national standards.

Emissions from three eight-day wildfire events were evaluated in the Interior Columbia River Basin Assessment and compared to prescribed fire emissions. Modeling with CALPUFF and photographs taken every six hours were used to compare smoke and ground level PM-10 concentrations. There were no violation of standards with prescribed fire, while wildfire resulted in standards violations. A 50 percent increase in PM-10 emissions from wildfires vs. prescribed fire was detected on a watershed basis (Ottmar 1995). However, "nuisance" conditions, including complaints from the public over visibility impairment, smoke odors or health concerns can occur at a much lower concentration of particulates than the national standards.

Researchers in the Pacific Northwest are quantifying how prescribed fire can offset emissions from catastrophic wildfire. A recent study in northeast Oregon in the Grand Ronde River Basin compared wildfire and prescribed fire emissions over a 4,856 km^2 (1.2 million acre) area. Six levels of prescribed fire treatment were simulated over a period of 100 years. Preliminary results indicated that total emissions would increase for the initial 30 years and then reach a plateau. The minimum level of emissions occurred when two percent of total acres were treated with prescribed fire (CH_2MHILL 1996). It was estimated that total emissions from fire can be reduced by 50 to 80 years of prescribed fire, even if only a

small amount of mechanical treatment to remove biomass is utilized (Ken Snell, pers. comm.). The Aldo Leopold Wilderness Research Institute, Missoula, MT, is preparing a long-term research agenda for wilderness fire that includes understanding, maintaining and restoring fire in ecosystems. Their scoping process for research needs is in progress.

The future of prescribed fire programs relies in part on quantifying how prescribed fire can offset wildfire emissions. In Oregon regulators are considering a "no net increase in emissions" policy based on that trade-off principle. Economic considerations should be driven by ecosystem management objectives, rather than least-cost techniques. The current federal policy is to "incorporate commodity, non-commodity and social values" in fire management programs (USDI and USDA 1995). By optimizing managed ignitions over short time periods, focusing on the removal of smaller organic matter while maintaining larger organic matter *in situ*, and applying state-of-the-art control technologies we can achieve both clean air and healthy ecosystems.

Smoke Modeling

The viability of a specific burn is often estimated from models to evaluate dispersion and estimate pollutant loads. Modeling is also important to quantify the total emissions contributed from prescribed fire in comprehensive emission inventories. Models are available that can simulate the behavior of smoke released from a fire. These range from relatively simple to rather complex in terms of how realistically they do the simulation.

Models are based on a rule of turbulent systems that establishes that it is impossible to predict the detailed behavior of a dispersing smoke plume. If it were possible to release smoke from the same place in the same turbulent atmosphere 100 different times, the smoke would go to 100 different places. However, over a large number of repetitions or 'realizations' there results an average called the 'ensemble average' condition. Since it is not possible to create strict ensemble conditions, modelers accept a second rule of turbulent atmospheric behavior that it is generally possible to replace an ensemble average with an average over time.

Models attempt to simulate time averages and their abilities improve as the averaging time lengthens. Thus, a model that attempts to predict the annual average concentration of smoke at a point has more chance for success than one attempting to predict a daily average. So the first, and perhaps most important aspect of modeling is to realize that it is unable to predict exactly where and how much smoke will result at any particular place and at any particular time. However, if one is willing to accept average conditions, our abilities increase as the averaging time frame of the prediction increases. It is more accurate to predict a monthly average concentration from a source than an hourly average, all other things being equal. In air quality modeling, a few approaches have developed to predict these average conditions. A class of important models are based on the assumption that the distribution of pollution at any arbitrary point downwind from the source is Gaussian, or bell-shaped. When this assumption is invoked, then the smoke is modeled by the centerline concentration and the width of the distribution. Thus, simulating the three dimension cloud of pollution is done by only three numbers, the centerline concentration and the widths in the two cross wind dimensions (parallel with, and perpendicular to the ground, for example).

A second simplification is with the nature of the wind speed and direction, the wind distribution that the model uses to carry the smoke. In nature this wind distribution is constantly changing and exhibits a complex pattern in three spatial dimensions. The simplest simulation, however, is to assume it does not vary spatially and therefore wind can be characterized by measurement, at a single location of its speed and direction, over time. The assumption made in the model is that this measurement is representative of the three dimensional distribution and can be used for characterizing the smoke dispersion. A good description of this concept is that the smoke acts like a flashlight, directed from a fire to a receptor. The validity of such an assumption is clearly dependent on the location, the presence of significant terrain features, the presence of significant weather conditions and the like. Again, the value of using a single point wind measurement improves if the measurement is taken over a year and used to make annual predictions of pollution impact. This approach is often used by the air pollution community to determine if a new source will exceed a standard level of pollution concentration and to develop levels of emissions controls for the source.

Clearly, with smoke there are additional factors requiring consideration. Fire moves, it consumes fuel and moves toward the fuel source. Fire is both

temporally and spatially complex, occurring in new and different locations all the time. Finally, fire is not released from a confined smokestack as are industrial pollutants: rather some smoke lingers in the forest canopy while some is lifted aloft. The fire intensity is not controlled like an industrial process so the amount of pollution and its energy of release varies. Hence, a third simplification is needed to deal with emissions produced by the fire. A model is needed to characterize the production of pollutants as well as the energy of pollutant release (to determine how high into the atmosphere a smoke plume may project, among other things) all as a function of time, as a function of fire type, fuel type, fire intensity, topography, fuel moisture content, and a host of other important variables.

Over the past 20 years we have succeeded in developing models based on the approximations described above. These models have been applied to provide estimates of smoke that might result from planned burning. They have been useful for planning purposes. However, we have also recognized that smoke does not always go where we estimate it should, especially from such simplistic models. For sensitive situations, those with especially sensitive populations or with greater-than-average complexity to the terrain and meteorology, we have been developing a more complex set of tools to predict smoke. One of the more significant improvements is in the air pollution model which allows pollution to be transported as discrete "puffs" of smoke released into a time and spatially varying wind field. This has been accomplished by developing a model of the wind distribution itself, a wind field model. It is applied to determine where each sequentially released puff of smoke might go, based on this wind field. The wind field might be upgraded each hour if sufficient meteorological data are available. Then a specific location on the ground (a receptor) is hypothetically monitored to determine how much pollution got to it or deposited upon it (Fox et. al. 1987). Combining a wind field prediction along with a detailed fire and pollutant emissions estimate represents the state of science in smoke management today.

Available Technologies for Smoke Management

SASEM.—The simple approach smoke estimation model (SASEM) was developed based on the simplest of these approaches using a straight line Gaussian plume model and crude estimates of fire emissions (Riebau et al. 1988). This model assumes that meteorological conditions do not change, and that the atmospheric distributions can be estimated by point values. The emission component has a simple plume rise calculation and emission model that was originally designed for application to woody, brush and grass fuels in rangeland conditions in the Rocky Mountains of the United States.

EPM.—The emission production model (EPM) was developed for forest logging slash burning in the Pacific Northwest on the United States (Sandberg and Peterson, 1984). It is a more detailed emission estimator than SASEM. More recent updates have broadened the fuels it can address.

TSARS.—The Tiered Smoke/Air Resources System (TSARS) was developed by the Bureau of Land Management (Riebau et. al. 1992) as a procedure that allows a manager to combine appropriate models to screen smoke emissions and improve decision making for burns. It includes the above two methods linked together as a decision support tool.

TSARS PLUS.—TSARS plus is a recently developed advance that includes the features of TSARS but adds the capabilities of a much more refined meteorological and puff dispersion model (NUATMOS and CITPUFF) (Hummelwerk, 1995) to the system. We will address TSARS PLUS in a later section.

CALPUFF AIR QUALITY MODELING SYSTEM.—The CALPUFF Air Quality Modeling System consists of CALMET, a meteorological model which produces wind and micro-meteorological fields linked with CALPUFF, a non-steady-state puff dispersion model. The system is being developed by a contractor to the US and Australian EPA's (Scire et al. 1995 a, b).

CALMET and CALMET (NUATMOS) Meteorological Models.—CALMET consists of a diagnostic wind field module (DWM) and micro-meteorological modules for over water and overland boundary layers. Scientists at Monash University (Ross et al. 1995a) have developed an alternative version of CALMET (henceforth referred to as CALMET (NUATMOS)) which uses the NUATMOS wind field model (Ross et al. 1988 a,b, 1991, 1993) in place of the diagnostic wind field module. Use of NUATMOS provides a few enhancements to the model. For example, Monash University has recently incorpo-

rated a prognostic module for simulating regional drainage flows into NUATMOS. The resulting model, and the 4D-Data Assimilation Technique used for merging wind observations and prognostic model predictions, have been evaluated for the Tamar Valley Airshed Study (TVAS) in Australia (Ross et al., 1994 a, b, 1995 b).

Planning a Prescribed Burn

The TSARS Plus system provides an excellent background for conducting a burning project. TSARS Plus combines a wind field model, NUATMOS, a puff dispersion model (CITPUFF, an earlier version of CALPUFF), SASEM, a simple emission estimator and EPM, a more detailed emission estimator. The system is available to anyone from its developers (Michael Sestak and Al Riebau, National Biological Service, Fort Collins, CO). TSARS Plus requires a PC compatible microcomputer with the following minimum requirements MS-DOS 6.0 or higher operating system; 80486DX 50 MHz (math coprocessor); 8 MB RAM; SVGA color monitor, 3.5 in. HD disk drive; 350 MB hard drive with at least 30 MB free; and laser printer (compatible with Hewlett Packard Laserjet).

Burn Project.—SARS Plus considers the following elements to a burn project: topography or digital terrain files; fires; receptors; meteorological stations; meteorological station data; wind files. The first step in conducting a burn project involves defining and developing each of these elements. Once these are all defined and saved, then it is possible to create a specific burn simulation based on using and linking these elements. Customization comes from modifying these elements for particular conditions.

Terrain Data.—Currently TSARS Plus includes diskettes for each of the western United States. Each state comprises a separate diskette except California which includes two diskettes. Terrain files produce a regular grid of elevations every 30 minutes of latitude and longitude. This represents approximately 1 km by 1 km in most of the western US.

Receptor Definition.—Receptors are the locations where smoke concentrations will be calculated. They are defined simply by identifying a name and a specific latitude and longitude. Each burn is limited to no more than 10 receptors.

Fire Definition.—The definition screen asks for the following information with specific input boxes: *fire name*, a descriptive fire title; *emission model* (SASEM or EPM). If SASEM is selected there are a number of additional fields that are to be specified, i.e., *burn type*—piled or broadcast; *fuel type*—wood, sage or grass fire area—allowable range 1-9000 acres; *number of piles* (if pile burn)—allowable 1-999; *total fuel loading*—allowable range 0.5-1000 cubic feet; *fireline intensity*—allowable range 1-7500 BTU/foot/second; and *fire duration*—allowable from 0.2-24 hours.

If EPM is specified a region needs to be selected from the options of Eastern Oregon, Western Oregon, Eastern Washington, Eastern Cascades, Western Cascades or other (user defined); land ownership is selected from National Forest System, other public lands, or private land; predominant species-Douglas Fir, grass, hardwoods, mixed conifer or ponderosa pine; fuel moisture type—adjusted NFDRS, NFDRS, weighted samples; 1000 hour fuel moisture; fuel loading by fuel size class and specific loading per size class; duff depth; average slope; harvest date, snow off month; and the number of days since significant rain.

Defining Meteorological Stations.—In order to simulate a burn it is necessary to use meteorological information. This information comes either from a physical observation or from a larger scale analysis or simulation of the mesoscale and synoptic scale weather pattern. TSARS Plus allows the user to identify meteorological stations by name, latitude and longitude and then to input atmospheric "soundings", namely information about the wind speed and direction, the temperature, atmospheric stability and mixing structure, all as a function of height. These data then become input information for the NUATMOS model.

Determining Wind Fields.—TSARS Plus runs NUATMOS to generate wind fields as outputs from the model. The model is run automatically from the TSARS Plus system. The user defines the location of the application (e.g. maximum and minimum latitude and longitude) and the size of the analysis area (limited to three choices, 100 km, 200 km. or 300 km square). Meteorological stations that are within the selected location are displayed for user selection, and available meteorological information are reviewed and selected.

Customizing the Burn Project.—The next step, after all the initialization and setups have been done, is to define or customize the actual burn. First the user selects specific locations, types and times for their burns. Second, the user defines the nature of

outputs, graphics, etc. that are desired from the burn. The third step is to actually conduct the simulation.

Results.—Results are available in both tabular and graphical output forms. Figure 1 illustrates the location, topography, and 24 hour average concentrations for a hypothetical prescribed burn in Arizona. Tabular output from a hypothetical burn is shown in Table 1.

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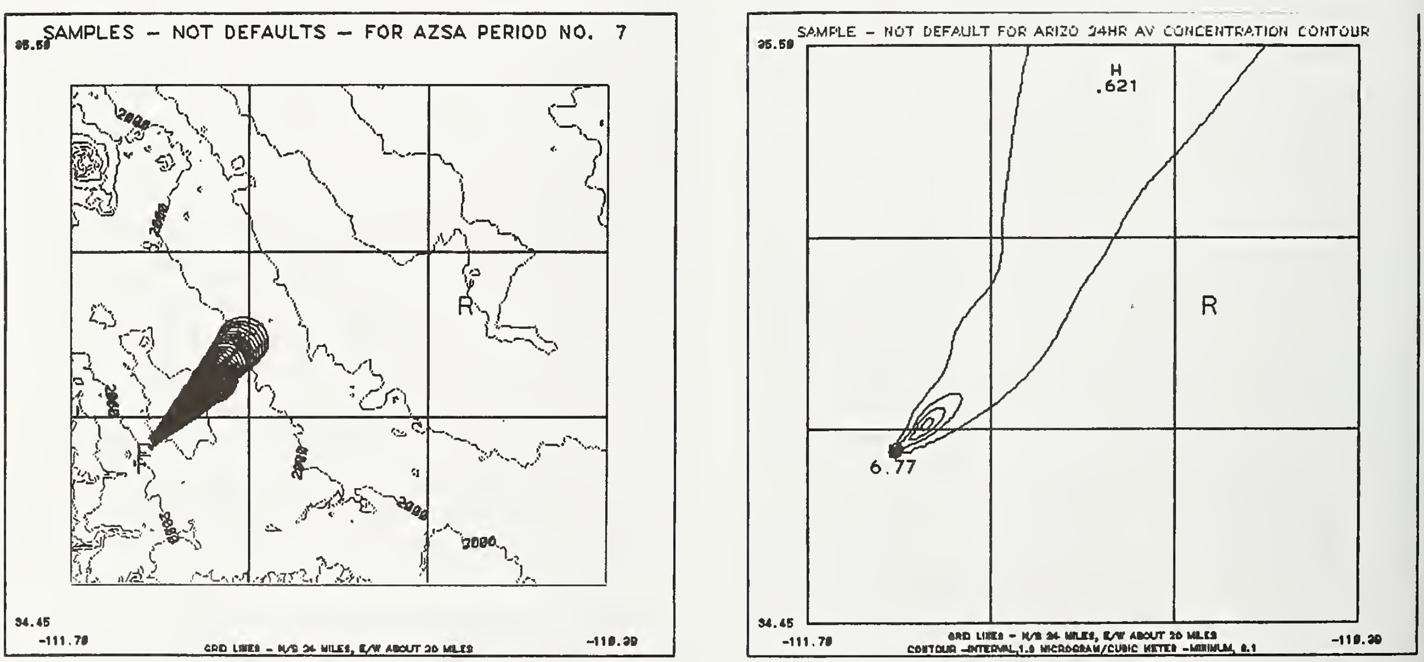

Figure 1. Sample TSARS Plus modeling output showing location, topography, and 24 hour average concentrations for a hypothetical prescribed burn in Arizona. Reproduced from the user's guide (Hummelwerk, Inc. 1995).

Table 1. Sample TSARS Plus model ouput for a hypothetical prescribed fire. Reproduced from the user's guide (Hummelwerk, Inc. 1995).

Concentrations on Plume Center Line for Each Fire

Burn Project Name: DENVER1
Title: COLORADO DENVER 40.2/105.3/1 1HR AV 1 SASEM FIRE
Beginning Date: 1995/08/24

Period	Fire ID	Puff Number	Concentration (ug/m**3)	Violation
1	1-FIRE1B	1	5.538400	NO
1	1-FIRE1B	2	12.880700	NO
1	1-FIRE1B	3	20.547500	NO
1	1-FIRE1B	4	25.439700	NO
1	1-FIRE1B	5	20.029100	NO
1	1-FIRE1B	6	6.347900	NO

1995/07/17

Predicted Hourly Average Concentration at Receptors
PM10 micrograms per cubic meter

Burn Project Name: VARIAB1
Title: SAMPLE SASEM variable wind field file
Beginning Date: 1995/07/14

Period	Receptor ID	Latitude	Longitude	Concentration PM 10 (ug/m**3)	Exceedance of Standard
1	1-RD14	43.200	107.000	0.000000	
1	2-SIMU2	43.250	109.200	0.000000	
1	3-SIMULATED	43.100	108.100	0.000000	
2	1-RD14	43.200	107.000	0.000000	
2	2-SIMU2	43.250	109.200	0.000000	
2	3-2IMULATED	43.100	108.100	0.000000	
3	1-RD14	43.200	107.000	0.000000	
3	2-SIMU2	43.250	109.200	0.000000	
3	3-SIMULATED	43.100	108.100	0.000000	
4	1-RD14	43.200	107.000	0.000000	
4	2-SIMU2	43.250	109.200	0.000000	
4	3-SIMULATED	43.100	108.100	0.000000	
5	1-RD14	43.200	107.000	0.000000	
5	2-SIMU2	43.250	109.200	0.000000	
5	3-2IMULATED	43.100	108.100	0.000000	
6	1-RD14	43.200	107.000	0.000000	
6	2-SIMU2	43.250	109.200	0.000000	
6	3-SIMULATED	43.100	108.100	0.000000	
7	1-RD14	43.200	107.000	0.000000	
7	2-SIMU2	43.250	109.200	0.000000	
7	3-SIMULATED	43.100	108.100	0.000000	

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Use of Fire in the Future: Benefits, Concerns, Constraints

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Abstract.—Fire as a natural occurrence or applied as a management tool can have beneficial effects on the vegetative communities of the Madrean Province, including reducing fuel loads, preparing seedbeds, increasing herbaceous plant production, improving wildlife habitats, and changing hydrologic processes. Opportunities for the use of fire in the future to obtain these and other benefits are explored in this paper. Concerns and possible constraints are also considered.

INTRODUCTION

Fire is an integral part of the ecology of the vegetative communities in the Madrean Province. The effects of fire on these communities can be either beneficial or detrimental, depending largely upon the nature of the fire, characteristics of the fire site, and values of the natural resources affected by the fire. The use of fire in the future to obtain beneficial purposes are explored in this paper. Concerns and possible constraints are also considered.

BENEFITS OF FIRE

Three broadly-defined vegetative communities are considered in this paper - montane (mostly pine) forests, woodlands (primarily pinyon-juniper and Madrean oak woodlands), and desert shrub and grassland communities. Opportunities for the use of fire in these vegetative communities for beneficial purposes are considered, where appropriate, in terms of reducing fuel loads, disposing of slash, preparing seedbeds, thinning overstocked stands, increasing herbaceous plant production, improving wildlife habitats, changing hydrologic processes, and improving aesthetic environments. While the focus of this paper is placed largely on opportunities for the use of prescribed fire, controlled burning treatments, and vegetative-modifying and vegetative-replacing wildfire can also be important in obtaining these benefits.

Montane Forests

Fire has been and continues to be used wherever opportunities present themselves in montane forests to reduce fuel loads, dispose of slash after timber harvesting, prepare seedbeds for regeneration, thin forest overstocked stands, increase the production of herbaceous plants, and improve wildlife habitats. Fire can also change hydrologic processes of a site.

Reducing Fuel Loads

The importance of fire in reducing the flammability of montane forests through the consumption of excessive fuel loads is recognized throughout the southwestern United States and northern Mexico. However, knowledge of fire intensities to meet fuel reduction objectives and techniques for controlling fire intensities remains incomplete. It is likely that periodically prescribed fire could be necessary to maintain the achieved reductions in litter amounts.

Predictive equations are available to estimate the consumption of naturally occurring fuel loads from readily obtained variables. Work by Harrington (1987) indicates that magnitudes of fuel consumption can

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be estimated from knowledge of the moisture content of the humus layer and either pre-fire litter depth or forest stand density. These equations present managers with an ability to prescribe the amounts of fuel that could be consumed by prescribed fire or through unplanned ignition.

Disposing of Slash

Timber harvesting is not widespread in the Madrean ecosystems of southwestern United States at this time, but logging does occur in northern Mexico. The additional fuels created by timber harvesting or pre-commercial thinning are often a major concern to managers. Piling and burning of slash is a preferred fuel treatment in ponderosa pine forests of the southwestern United States (Hirsch et al. 1979). Analysis of fuel treatments must involve considerations of the effects of the treatments on future timber production, wildlife habitat, soil characteristics, water yield and quality, and recreational values.

Preparation of Seedbeds

Conditions affected through burning that favor the germination of seeds and early growth of seedlings include creation of a more "receptive" seedbed through the removal of litter accumulations and exposure of mineral soil, increased nutrient availability, and more favorable soil moisture and temperature regimes. Seedbeds can remain receptive to regeneration for several years if prescribed burning is repeated.

A seed crop is also required for successful natural regeneration. It is important, therefore, that potential seed trees not be severely damaged in burning treatments to prepare a seedbed. Fire-scorched trees (as opposed to trees whose foliage has been totally consumed) can have a relatively high rate of survival, depending largely upon the time and intensity of the fire and extent of scorching.

Ponderosa pine seedlings have been successfully planted after fire has removed competing vegetation and, in doing so, preparing the sites for planting.

Thinning Overstocked Stands

The use of fire to thin overstocked and often stagnated forest stands has been prescribed by managers in some situations. The effectiveness of fire in thinning of overstocked stands is largely related to the concentrations of fuels on the ground before burning. While burning can be a "good start" in reducing the density of overstocked stands, it is likely that additional prescribed fire will be necessary before these stands reach optimal productivity.

Attention must also be directed to the survival of fire-damaged trees that remain in the stands. Only rarely will all trees escape damaged by fire. Therefore, survival potentials of fire-damaged trees should be incorporated into the planning of prescribed fire to thin stands.

Increasing Herbaceous Plant Production

Herbaceous plant growth can be increased through burning. Competing forest overstories are often reduced in density and soil fertility is usually increased by the release of nutrients, encouraging the growth of herbage plants. Plant vigor is promoted by the removal of senescent shoots and foliage and, in many cases, burning of the litter prevents the interception of light and water, once again, favoring plant growth. Because of the wide variation in the responses of herbage plants, however, generalizations on the effects of fire on herbage production should not be made.

Nutrient values of herbaceous plants, including crude protein, phosphorus, and in vitro digestibility are often enhanced (at least temporarily) by burning (Pearson et al. 1972).

Improving Wildlife Habitats

Food and cover for wildlife, both game and nongame species, can be modified by the disturbance of fire. That these changes impact on wildlife habitats in montane forests, often beneficially, has been generally recognized.

The effects of fire on wildlife habitats should be evaluated only in the long-run in many instances; it is often that changes in habitat conditions from fire are "felt" only years later. Depending largely upon the fire intensity, season when the fire occurred, and type of habitat burned, deer summer-fall use of a site declines in the first few years after burning, only to increase in later years, with the "net effect" of the fire being an improvement in habitat conditions (Lowe et al., 1978). Conversely, habitats for tree-foliage-searching birds can increase in numbers in the first years

after a fire, and then decrease to below pre-fire levels for the next several years and remain there into the future. It is important, therefore, that opportunities to use fire in improving the habitats for wildlife species be considered in the long-term and on a "case by case" basis.

Changing Hydrologic Processes

Fire which reduces litter depths and the density of a montane forest stand can make more soil water available on a site, which (in turn) might make more water available for overland flow. Average runoff efficiencies, that is, the ratio of surface runoff to precipitation, generally increase with the severity of burn (Campbell et al. 1977).

Sediment concentrations and the chemical quality of streamflow regimes after fire are also important to managers. The exposure of soil and subsequent soil movement can increase by "light" and "moderate" burning, although much of the eroded soil materials only moves a short distance downslope and becomes stabilized shortly after the burning in many instances.

Concentrations of chemicals (calcium, magnesium, fluoride, etc.) in streamflow can increase after burning (Sims et al. 1981). However, these elevated concentrations often decline in subsequent runoff events.

Improving Aesthetic Environments

Controlled burns can improve the aesthetics of a montane forest by keeping the forest open and "park-like," and favoring the development of large, individual trees that can be readily viewed by the public. Monotonous, dense, debris-litter clumps of saplings are generally uninviting both visually and physically. The adverse scenic impacts of fire are frequently short-lived and, therefore, might not be a significant deterrent to prescribed burning in similar conditions.

Woodlands

There is little information on the use of fire in the pinyon-juniper or Madrean oak woodlands in reducing fuel loads, although the conversion of these woodland overstories to a dominance of herbaceous plants can result in significant increases in herbaceous fuels. Fire can increase the production of herbaceous plants in pinyon-juniper woodlands and, to a lesser extent, the Madrean oak woodlands. Properly planned fire can also improve wildlife habitats and change hydrologic processes in these woodland ecosystems.

Reducing Fuel Loads

Ground fuels, rarely heavy in pinyon-juniper and Madrean oak woodlands, are often consumed by burns occurring on flat to gently rolling terrain. However, the continuity of fuel loads on rougher terrain can be insufficient to support fire spread.

It is important that planned fire "hot enough" to significantly reduce fuel loads not kill the trees at the same time, unless their removal is also an objective of the burning.

Increasing Herbaceous Plant Production

The production of grasses, grass-like plants, and forbs is generally increased with a decrease in densities of pinyon-juniper overstories, while little change has been observed in Madrean oak woodlands. Therefore, production of herbaceous plants can be increased by burning (often with seeding treatments) in pinyon-juniper woodlands.

The time of burning is important in terms of its influence on herbaceous plants. Winter burning appears to have little effect on the density or vigor of forage species, unlike the results reported for warm-season burns (Pase 1971). Winter burning might be helpful in maintaining a dominance of herbaceous plants in woodland areas converted, however.

Another benefit of burning can be a release of nutrients for subsequent plant growth.

Improving Wildlife Habitats

When burned areas are kept small and interspersed with trees, an increase in both food and protective cover for a variety of wildlife species is likely to occur. Furthermore, the edge effect created by the burned openings generally enhances wildlife environments.

Small burns within unburned woodlands, for example, create a greater variety of food and cover for deer than is available on large areas, either burned or unburned. Opening up the dense overstories by fire provides additional space for deer movements, and an increase in the abundance and quality of browse

species. Increases in quail and other bird populations are also frequently noted after the burning treatments.

Changing Hydrologic Processes

Conversion of pinyon-juniper and Madrean woodlands to herbaceous plants should reduce the loss of water to consumptive use and, in doing so, increase streamflow. Unfortunately, streamflow response to fire has not been measured adequately to verify this point.

Movement of eroded soil materials immediately after a fire can be greater than in later years. As a consequence, increases in sediment yields can also occur in the short-run. Surface runoff from burned watersheds subsequently treated with herbicides can become contaminated by the herbicides.

Desert Shrub and Grassland Communities

The use of fire in desert shrub and grassland communities has been largely confined to the control of shrubs and conversion to perennial grasses and forbs. Such conversion treatments can improve the habitats for wildlife species. Opportunities to use fire as a management tool can be limited, however, because the fuel loads are often inadequate to support burning.

Increasing Herbaceous Plant Production

While the production of herbaceous plants might be reduced by burning treatments, many of the plants recover by the second growing season after the fire (Cable 1967). It is likely, therefore, that a "temporary" reduction in the production of herbaceous vegetation has little permanent consequences. Abundance of some plant species can depend upon episodic fire in periods of reduced precipitation (Bock et al. 1995).

Periodic fire can be used to maintain increases in the production of forage species at the expense of shrubs.

Improving Wildlife Habitats

Fire can improve the habitats for deer in desert shrub and grassland communities. Recurring fire that burns in irregular patterns, leaving unburned patches, often creates a diversity of habitat conditions.

The habitats for bird populations are also improved by fire in many instances (Bock et al. 1976). This response to fire is attributed to the floristic changes after the fire, resulting in an increase in the diversity and productivity of the birds.

CONCERNS AND CONSTRAINTS

There are concerns and possible constraints that must be addressed before burning programs become more operational in the Madrean Province ecosystems. Environmental factors, economic considerations, and a need for public support are all important in this respect.

Environmental Factors

The use of fire on sites where fuels have accumulated for many years can result in changes in surface organic materials and nutrient storage. In a study in a ponderosa pine forest, Covington and Sackett (1984) reported that the storage of surface organic materials was reduced by a prescribed fire, with nutrient storage somewhat less affected. The burning also released much of the nutrients bound in the surface organic materials, however, improving the conditions for plant growth. It is important, therefore, that guidelines for prescribed burning consider the impacts of fire on nutrient regimes and, as a result, the potential loss of nutrients.

Burning of organic materials on the soil surface removes the protective litter layers, volatilizes large amounts of nitrogen and smaller amounts of other elements, and transforms less volatile elements into soluble forms that are more easily absorbed by plants or lost by leaching. Heating of the underlying soil layers also alters the physical, chemical, and biological properties of the soil that are dependent upon the organic materials on the soil surface. These effects are variable and largely unpredictable.

All mineral soils containing small amounts of organic materials are likely to become water repellent, to some degree, when heated (DeBano 1981). The severity and distribution of the water repellency after a fire will determine the subsequent management problems on the site. Prescribed fire can be a

practical method of modifying water repellency by controlling the occurrence and behavior of the fire.

The use of fire on a widespread-scale can be constrained by smoke. Smoke management to minimize the impairment of air quality includes fuel management and fire prescriptions that improve combustion efficiencies; firing and "mopping up" techniques to reduce emissions; and the scheduling of burning to enhance convection and dispersion, and to ensure plume trajectories away from "sensitive" areas.

Economic Considerations

Althaus and Mills (1982) indicate that minimization of the fire cost, plus the net change in natural resource outputs, that is, C + NVC, is the appropriate selection criteria for analyzing the economic efficiencies among alternative fire management strategies. The C + NVC criterion considers both beneficial and detrimental effects of fire. However, site-specific data must be available to quantify costs and the changes in the natural resource values for the alternative fire management strategies considered.

Information on the costs of burning in Madrean Province ecosystems is scarce. This lack of information might be attributed to deficiencies in reporting procedures. To evaluate the economic efficiencies of alternative fire management strategies, therefore, additional cost information representing a range of conditions and sites is necessary.

Changes in natural resource values should consider the effects of fire on both on-site and off-site values. Natural resources valued in monetary terms should be included in calculations of net value change; but, values that cannot be measured in monetary terms should not be "forced" into an analysis of economic efficiencies. It is also important to evaluate the changes in natural resource values in the long-run, as these values can drastically change over a period of time after a fire.

Public Support

Opportunities to use fire in the future for beneficial purposes are largely dependent upon public support. That the public recognizes that fire can be both beneficial and detrimental has been shown by Cortner et al. (1984), who also determined that public acceptance and understanding of the potential benefits of fire are often high, in relative terms. Furthermore, knowledge of the effects of fire on natural resources can increase the "tolerance" for fire by the public (Taylor et al. 1986). To be effective in obtaining public support for fire management programs in the future, educational efforts should be oriented to the local conditions to be affected, and to local knowledge and acceptance of fire.

CONCLUSIONS

Fire has a role in the management of Madrean Province ecosystems. Depending largely upon the vegetative community, fire offers opportunities for reducing fuel loads, disposing of slash, preparing seedbeds, thinning stands, increasing herbaceous plant production, improving wildlife habitats, changing hydrologic processes, and improving aesthetic environments. At the same time, however, there are environmental, economic, and educational concerns and constraints to the use of fire. Benefits of fire, therefore, must be reconciled with these concerns and constraints before fire becomes a widely applied management tool in the future.

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The Let-Burn Policy: Implications in the Madrean Province of the Southwestern United States

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Abstract.—Fire is a natural phenomenon in Madrean Province ecosystems. Suppression of natural fire has altered ecosystem processes, however. Recognition of these alterations has led to the adoption of let-burn policies by many management agencies, but a let-burn policy has become less viable in recent years in the opinion of many people. There are barriers to a let-burn policy that should be weighed against the possible benefits before deciding upon the proper course of action relative to acceptance or rejection of the policy.

INTRODUCTION

Fire is a natural phenomenon in Madrean Province ecosystems. Suppression of natural fire has drastically altered ecosystem processes, however, threatening fire-adapted plant and wildlife communities. Recognition of these alterations has led to the adoption of tenuous let-burn policies by many management agencies, but a let-burn policy has become less viable in recent years in the opinion of many people. There are barriers and benefits to a let-burn policy in the Madrean Province of the southwestern United States, some of which are reviewed in this paper.

A few terms should be clarified at the outset. *Natural fire* refers to fire that begins by natural ignition (usually lightning). A *wildfire* is out of control of human suppression efforts, regardless of ignition source. Prescribed fire is human-ignited fire that is under control of human suppression efforts. The term *prescribed fire* has been used elsewhere with no distinction of the source of ignition, but rather to indicate the existence of conditions that are acceptable for burning, as deemed by the appropriate management authority. A *let-burn policy* is that which mandates the allowance of natural fire to burn, usually within some prescription of fuel and weather characteristics. Let-burn policy is the focus of this paper.

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BARRIERS

There are barriers (real and perceived) to acceptance of a let-burn policy, not only in the Madrean Province but more generally. Among these barriers are:

- *Past suppression efforts*—That management agencies have a difficult time in subscribing to a let-burn philosophy is, ironically, a direct outcome of earlier suppression philosophies. While the role of natural fire in ecosystem management is sound conceptually, in practice it is mired in a paradox—long-term protection from fire has altered vegetation and fuels in many natural ecosystems, increasing the risk of severe wildlife and reduced resource values (Arno and Brown 1991).
- *Conflict of interest*—It is often the inclination (conscious or not) of managers to attack a fire, although many also recognize potential long-term benefits of burning. This inclination is rooted in traditional and budgetary incentives. Many managers have, or even exercise regardless of having, decisionmaking authority on fire suppression.
- *Concern for neighboring landowners*—There are a number of reasons, based largely on how a let-burn decision could affect neighboring landowners, why a manager might decide in favor of fire suppression; these reasons include water

quality effects, reduced tourism, loss of income and jobs, smoke, and public safety. A management agency would logically have concerns about each of these. Even if it did not, agencies area often forced into having these concerns out of fear of liability and litigation.

- *Political pressures*—Political pressures can be brought to bear on agency officials responsible for fire suppression. Although there was a country-wide political trend toward acceptance of natural fire as ecologically appropriate, the Yellowstone fires of 1988 marked a turning point in this evolution (Schullery 1989). These and other catastrophic wildfires in recent years have generated a dialogue that goes beyond the issue of fire in national parks and forests to more fundamental questions about the management of public lands.

- *Expanding urban interfaces*—The desire for a rustic home in a quasi-rural setting is problematic, considering the occurrence of fire in natural ecosystems. Not only are these homes in areas where the possibility of fire remains prominent, but it is frequent that the houses and their yards are fuel-rich themselves (Beebe and Omi 1993). Therefore, when faced with a fire-suppression decision, a manager has to consider this urban interface as conducive to property damage and risky to human life.

- *Uninformed public*—Much of the pressure that stymies management agencies in allowing more natural fire to burn stems from an uninformed public—the public is often a philosophical step behind the state of natural fire science. While Cortner et al. (1990) gave a "passing grade" to regional publics for their knowledge of fire effects and management policies, much of the general public is still in the Smoky Bear school, i.e., that fire is bad. Even the agency that spawned the Smoky phenomenon is aware of this problem.

- *Smoke concerns*—Aside from (but probably related to) the physical effects of smoke on human health (with people reporting nausea, headaches, and other symptoms that they attributed to a fire's smoke), smoke is a psychological "red flag" to people residing in the area of a fire (Easthouse 1993). In addition to their concerns for the public welfare, management agencies must comply with the Clean Air Act. This compliance can be a major obstacle to let-burn policy.

- *Economics of scale*—Economics of scale are a practical and logical obstacle to let-burn policy. When natural ignition occurs, there is often an option of putting the fire out quickly, with a relatively small investment of time and money. However, if the fire is allowed to burn under prescription, but turns into a wildfire that falls outside of prescription, it can turn into a fiscal sink.

BENEFITS

There can be benefits to allowing natural fire to burn in montane (mostly pine) forests, pinyon-juniper and Madrean oak woodlands, and desert shrub and grassland communities. These benefits should be properly weighed against the barriers of a let-burn policy before deciding upon the proper course of action regarding the policy.

- *Reducingfuel loads*—That fire reduces the flammability of montane forests and pinyon-juniper and Madrean oak woodlands through the consumption of excessive fuel loads is recognized in the Madrean Province (Ffolliott 1990). One effect of reducing fuel loads by allowing fire to burn can be a decrease in the frequency of large, devastating wildfires.

- *Preparation of seedbeds*—Conditions affected through burning that favor the germination of seeds and early growth of seedlings in montane forests include creation of a more "receptive" seedbed because of the removal of litter accumulations and exposure of mineral soil, increased nutrient availability, and more favorable soil moisture and temperature. Sprouting of tree and shrub species commonly found in pinyon-juniper and Madrean oak woodlands can be stimulated by fire.

- *Thinning stagnated stands*—The effectiveness of natural fire in thinning of stagnated and overstocked montane forest stands is largely related to concentrations of fuels on the ground. Potential "crop trees" and, in general, trees of high multiple-use value can be released from understory stands, as the smallest trees are killed.

- *Increasing herbaceous plant growth*—Herbaceous plant growth can be increased through burning. Soil fertility can be increased by the release of nutrients, encouraging growth of plants (Pearson et al. 1972). Plant vigor is promoted by the removal of senescent shoots and foliage and, in many instances, burning of the litter prevents the interception of light and water, favoring plant growth. Crude protein, phosphorus, and *in vitro* digestibility of herbaceous plants are often enhanced (at least temporarily) after burning.
- *Improving wildlife habitats*—The environment in general is modified by the disturbance of fire. These changes often improve habitats for deer, quail, and other bird populations in the Madrean Province (Bock et al. 1976). This response to fire is attributed to floristic changes, increasing plant productivity and diversity.
- *Changing hydrologic processes*—Fire that reduces litter depth and the density of montane forests can make more soil water available on a site, which (in turn) can increase overland water flows (Campbell et al. 1977). Sediments, and calcium, magnesium, and other chemicals in streamflow can increase after burning, although these elevated concentrations usually decline in subsequent runoff events.
- *Improving aesthetic environments*—Natural burns can improve the aesthetics of montane forest ecosystems by keeping the forests open and "park-like," and favoring the development of large, individual trees that can be viewed readily by the public. Monotonous, dense, debris-littered clumps of saplings are generally uninviting both visually and physically. Adverse scenic impacts of natural fire are often short-lived and, therefore, might not be a significant deterrent to prescribed burning in similar conditions.

CONCLUSIONS

Barriers to a let-burn policy in the Madrean Province of the southwestern United States should be weighed against the possible benefits before deciding upon the proper course of action relative to acceptance or rejection of the policy. It is our opinion that a let-burn policy of some form is essential to the management of ecosystems in the region. Such policy could be promoted through public education, incentives for appropriate let-burn decisions, prescribed fire to reduce fuel loading problems, and zoning regulations.

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Initial Assessment of Fire-Damaged Mesquite Trees Following an Illegal Burn

Leonard F. DeBano[1], Gerald J. Gottfried[2], Jose Villanueva-Diaz[1], Peter F. Ffolliott[1], and Diego Valdez-Zamudio[1]

Abstract.—Effects of an illegal burn on the Santa Rita Experimental Range on the mesquite (*Prosopis velutina*) component of the ecosystem were assessed in the late fall of 1995, nearly 18 months after the fire. While most of the mesquite were damaged by the fire, stocking decreased only a 10 percent compared to pre-burn stocking conditions. It is possible, therefore, that stocking by mesquite will return to pre-burn conditions in the absence of management practices to modify the occurrence of mesquite on the site.

INTRODUCTION

An illegal fire was set in a mesquite-dominated semidesert grassland in Sawmill Canyon on the Santa Rita Experimental Range, located 30 miles south of Tucson, Arizona, in the early summer of 1994. This fire, during the peak of the "high fire danger" season for the year, burned approximately 80 acres. The effects of the burn on the mesquite component of the ecosystem were assessed in the late fall of 1995, nearly 18 months after the fire. The results, presented here, add to the literature-base on the effects of fire on mesquite in semidesert ecosystems of the northern Madrean Province (Reynolds and Bohning 1956, Cable 1965, 1967, Alonso 1967, White 1969, McLaughlin and Bowers 1982, Cox et al. 1990).

SITE OF THE FIRE

A general description of the Santa Rita Experimental Range is found in the literature (Martin and Reynolds 1973) and, therefore, need not be repeated here. The fire burned on a rocky site about 1-1/2 miles from the Florida Canyon Headquarters, near the southern boundary of the Experimental Range.

The site is about 3,900 feet in elevation, on largely southerly slopes ranging from 5 to 20 percent, extending to an adjacent ridgetop. The overstory before the fire was dominantly mesquite with ocotillo (*Fouquieria splendens*) scattered throughout. The perennial grass component is largely Lehmann lovegrass (*Eragrostis lehmanniana*). Black grama (*Bouteloua eriopoda*) and Arizona cottontop (*Digitaria californica*) are also found intermixed. The amount of fine fuels and accumulation of organic material at the base of trees before the burn are not known.

METHODS

Numbers of fire-damaged mesquite and mesquite with no visible damage were counted on sixty 1/20-acre sample plots located at 100-foot intervals along a series of temporary transects. Mesquite trees tallied on the plots were classified with respect to fire damage as follows:

- No visible damage.
- Scorched crowns (partial crown kill), in which case the occurrence or absence of sprouting was noted.
- Shoot-killed (entire crown killed, but sprouting).
- Root-killed (entire crown killed, no sprouting).

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Diameter at root collar (drc) was measured on each mesquite tree tallied. Equivalent diameter at root collar (edrc) was subsequently calculated for multiple-stemmed mesquite (Batcheler 1985, Chojnacky 1988). EDRC is the square root of the sum of drc values of the individual stems.

RESULTS AND DISCUSSION

A total of 257 mesquite trees (equivalent to 85.7 trees per acre) were tallied on 37 (61.7 percent) of the 60 sample plots, providing a measure of pre-burn stocking conditions on the site. The remaining sample plots were not stocked with mesquite.

The percent of mesquite tallied in the fire-damage classes is shown in figure 1. Less than 15 percent of the trees had partially scorched crowns, more the one-half were shoot-killed, and about one-third were root-killed. Less than 5 percent of the trees showed no visible damage.

There was no significant relationship between the occurrence of mesquite trees in the fire-damage classes and their respective diameters (drc or edrc), nor between the occurrences of these trees and the number of trees tallied on the sample plots. Such relationships (or the lack thereof) are considered to be fire-specific, as they have been reported in some form in other studies on the effects of fire on mesquite trees (Reynolds and Bohning 1956, Cable 1965, 1967, White 1969, McLaughlin and Bowers 1982).

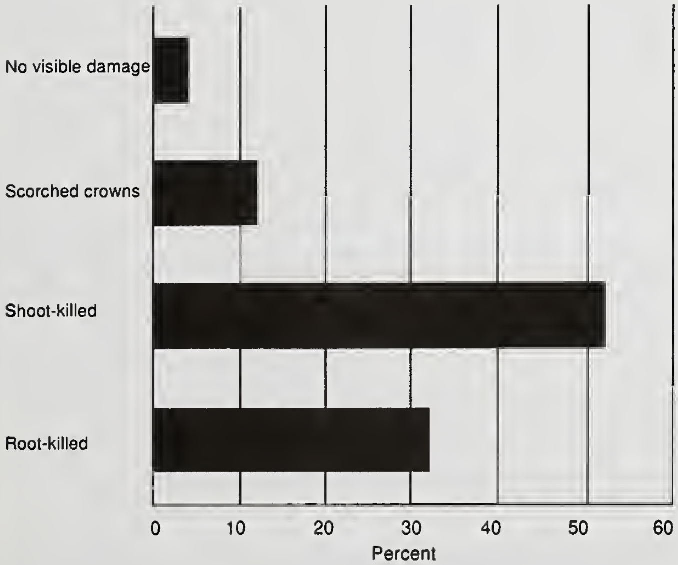

Figure 1. Percent of mesquite trees in fire-damage classes.

Partially scorched mesquite often recovers by sprouting from either basal stem buds below the ground surface or auxiliary buds on the branches of the crowns (Cable 1965, White 1969, McLaughlin and Bowers 1982). To determined whether this happened here, we compared the occurrence of sprouting in mesquite trees with no visible damage to that of mesquite with scorched crowns; the occurrence of sprouting in trees with no visible damage was assumed to be representative of the pre-fire situation. Shoot-killed and root-killed mesquite trees were not included in the comparison.

	Sprouting (percent)	No sprouting (percent)
No visible damage	27 (3)	73 (8)
Scorched crowns	71 (22)	29 (9)

While the sample of trees is small (the number of trees is presented in parenthesis), the increase in sprouting after the fire is consistent with that observed by Cable (1965), White (1969), and McLaughlin and Bowers (1982). Basal sprouts immediately below the ground surface were the most commonly observed form of sprouting in the scorched trees. Basal sprouts were also common in shoot-killed mesquite.

MANAGEMENT INFERENCES

Most of the mesquite on the burned site were damaged by the fire in some way. However, many of the fire-damaged trees subsequently sprouted in the 18 months since the burn. Live mesquite trees, sprouts, or both were tallied on 31 (51.7 percent) of the 60 sample plots, representing only a 10 percent decrease in stocking when compared to the assumed pre-burn stocking conditions. It is possible, therefore, that stocking by mesquite will return to the pre-burn conditions in the absence of management practices to prevent their return.

Research is needed to determine the effect of repeated prescribed burning on mesquite sprout mortality and the build-up of fine fuels.

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Properties of Forest Soils on Mountains of New Mexico, USA and Sonora, Mexico

Jose Villanueva-Diaz and Guy R. McPherson[1]

Abstract.—Physical and chemical properties of soils associated with forest communities were described. We hypothesized the differences in land-use practices (e.g., grazing, logging, fire suppression) have produced long-term effects in nutrient availability and other soil properties, in mountains on either side of US/Mexico border. Soil depth, litter depth, OM, total nitrogen, CEC, and most soluble and exchangeable ions were significantly affected by the interaction mountain range-forest community ($p < 0.05$). Long-term effect of land use on soil properties apparently was masked by other environmental factors.

INTRODUCTION

The study of patterns and rates of nutrient circulation is of vital interest for understanding and managing forest ecosystems. The ability of soils to provide available plant nutrients is determined not only by the relative concentrations of elements in the various soil fractions and soil solution but also by the rates that these elements are released to the soil. Under natural conditions fire plays an important role in nutrient cycling by releasing nutrients that otherwise would be tightly tied to organic materials for long periods of time.

Considerable effort has been devoted understanding the effect of wild fires and prescribed burns on soil nutrients immediately or for several years following fire events. However, no studies have addressed the morpho-chemical composition of soils supporting forest communities subjected to different land-use practices (e.g., grazing, fire exclusion, and logging).

The objectives of this study were to compare morphological and chemical composition of soils:

- At different elevations within a given mountain range, and
- Between mountain ranges having similar plant communities, but having different land use.

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STUDY AREAS

Animas Mountains and Sierra los Ajos

The Animas Mountains (AM) are the highest range in southwestern New Mexico, west of the Rio Grande and south of the Mogollon Plateau (31° 35' N latitude, 108° 47' W longitude). The mountain range rises to 2600 meters and extend over a 100 square-kilometer area on the Gray Ranch of southern Hidalgo County, NM. (Fig. 1).

The AM have a bimodal precipitation pattern with about 60% of the average annual precipitation (450-750 mm, depending on elevation) occurring in July-September and 40% received during the winter months. Temperatures above 32°C are common during the summer and usually range between 12°C and -5°C during the winter. Vegetation of the AM is composed of three main types: lower encinal, upper encinal, and forests (Wagner 1977).

Sierra Los Ajos (SLA) is located in Sonora, Mexico (30° 55' N latitude, 109° 55' W longitude). The enclosed area encompasses approximately 171 square kilometers. The SLA has a climate and vegetation similar to the nearby AM (Garza-Salazar 1993).

LAND-USE HISTORY

Land-use practice differences between the AM and the SLA exist with relation to fire suppression,

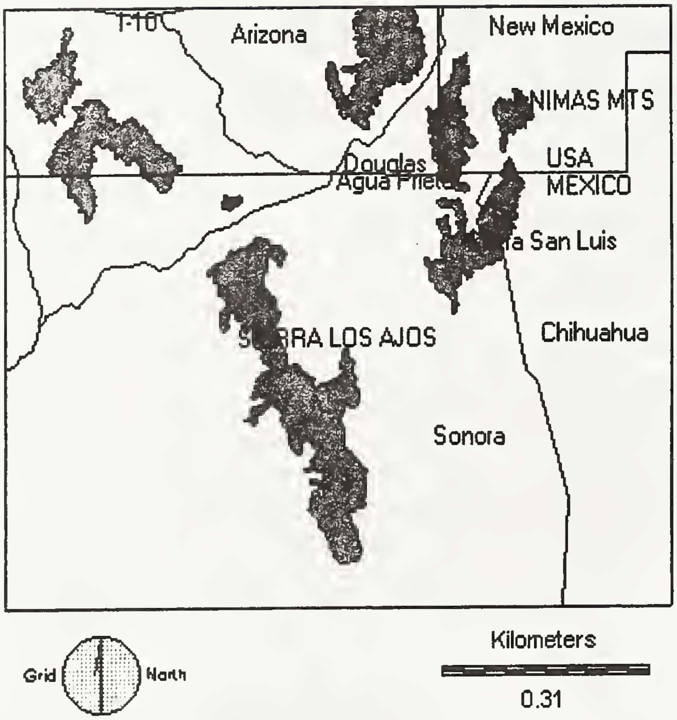


Figure 1. Geographical location of Animas Mountains New Mexico and Sierra los Ajos, Sonora.

intensity of grazing, and logging activities. A fire suppression policy was implemented for the AM and in general for southwestern USA at the beginning of the 20th century. On the other hand, the SLA has not been subject to complete fire suppression activities or at least not to the level of intensity provided for southwestern USA mountains.

Land-use history for both mountain ranges is described with more detail in companion paper "Effects of Climate, Fire, Land-Use History, and Structural Development on Forest Communities" in this proceedings.

METHODS

The present study considers three forest communities in each mountain range:

- Douglas-fir/gambel oak (Df/GO) forests found on northern aspects above 2200 m,
- Southwestern white pine/ponderosa pine/chihuahuan pine (MP) forest found at intermediate elevations, and
- Pinyon pine/juniper/oak (PJ) forests found at lower elevations.

During the summer of 1992 and 1993, four representative stands of each community in each mountain range were studied intensively (n= 24 stands).

One, 20×50 m (0.1 ha) permanent plot was established in each stand. Each plot was divided into ten, 10×10 m subplots. A representative site of each plot was selected and a pit was dug to expose a soil profile.

Soil surface determinations of litter depth (cm), duff depth (cm), ground cover (%), percentage of gravel (0.2–7.6 cm), cobble (7.6–25.4 cm), and stones (> 25.4 cm) were obtained in a 4 m^2 (×2m) subplot assigned randomly to each one of the plots. Average visual estimations of percentage of gravel, cobble, and stones were also obtained in the soil subsurface within the pit previously described.

From each subplot a soil sample from the first 20–25 cm depth was collected. In the field soil samples were passed through a 2 mm sieve. Approximately 1 kg of this sieved soil was stored in a plastic bag for morphological and chemical analysis. Morphological determinations consisted of particle-size analysis (% sand, silt, and clay), estimated with the hydrometer method; soil color components (hue, value, and chroma) were obtained according to procedure in Post et al. (1993); and bulk density (g/cm3) using the clod method (Blake and Hartge 1986).

Chemical determinations consisted of pH, measured on a soil/solution ratio of 1:2; electrical conductivity (mmhos/cm) was determined from saturation extract; total carbonates were analyzed using the volumetric technique; organic matter (%), using the Walkley-Black method (Walkley 1947); total nitrogen (%), using the Kjeldahl method; available phosphate (ppm) according to Olsen and Sommers (1982); soluble ions (me/l) from saturation extract; cation exchange capacity (me/100 g), following procedure described by Polemio and Rhoades (1977), and exchangeable cations (me/100 g) were extracted by ammonium saturation method. Chemical determinations were done mostly following procedures described in the 1982 Methods of Soil Analysis.

Statistical Analysis

An Analysis of variance (ANOVA) was used to evaluate the effect of independent variables (mountain range, communities, and interaction mountain range-forest community) on dependent variables (morphological and chemical determinations). A

Tukey's Studentized Range (HSD) Test was used to separate means for those significant variables (p <0.05).

Relationships between communities (elevation) and dependent variables were analyzed with Pearson's Product Moment (r).

RESULTS AND DISCUSSION

Interaction of Mountain Range with Forest Community

Soil depth, litter, organic matter, total nitrogen, pH, particle-size fractions, ground cover, cation exchange capacity, color (value and chroma), soluble ions (Na, K, SO4), and most of the exchangeable cations were significantly influenced by the interaction mountain range-forest community.

Soil depth significantly increased from lower (PJ) to higher elevation (Df/GO) communities at the AM. However, soil depth was not different for upper elevation communities (MP and Df/GO) at the SLA (Fig. 2). This situation could be explained considering that MP and Df/GO communities at the SLA usually were present at similar elevations.

Litter depth significantly increased along the altitudinal gradient for the AM and it was not significantly different for communities in the SLA. Forest fire suppression coupled with increased precipitation and reduction of temperature along the gradient, probably was responsible for the increase in the AM. On the other hand, repeated fires in the SLA may have reduced litter depth especially at upper elevation communities that usually are subjected to more frequent fires.

Although significantly correlated, organic matter (OM) and total nitrogen did not show a consistent increase trend with elevation in either mountain range (Fig 3. and Fig. 4). Greater concentrations of both OM and total nitrogen were found for MP and PJ communities in the AM. Comparatively, OM and total nitrogen were lower and fairly constant for the SLA communities. Covington and Sackett (1986)

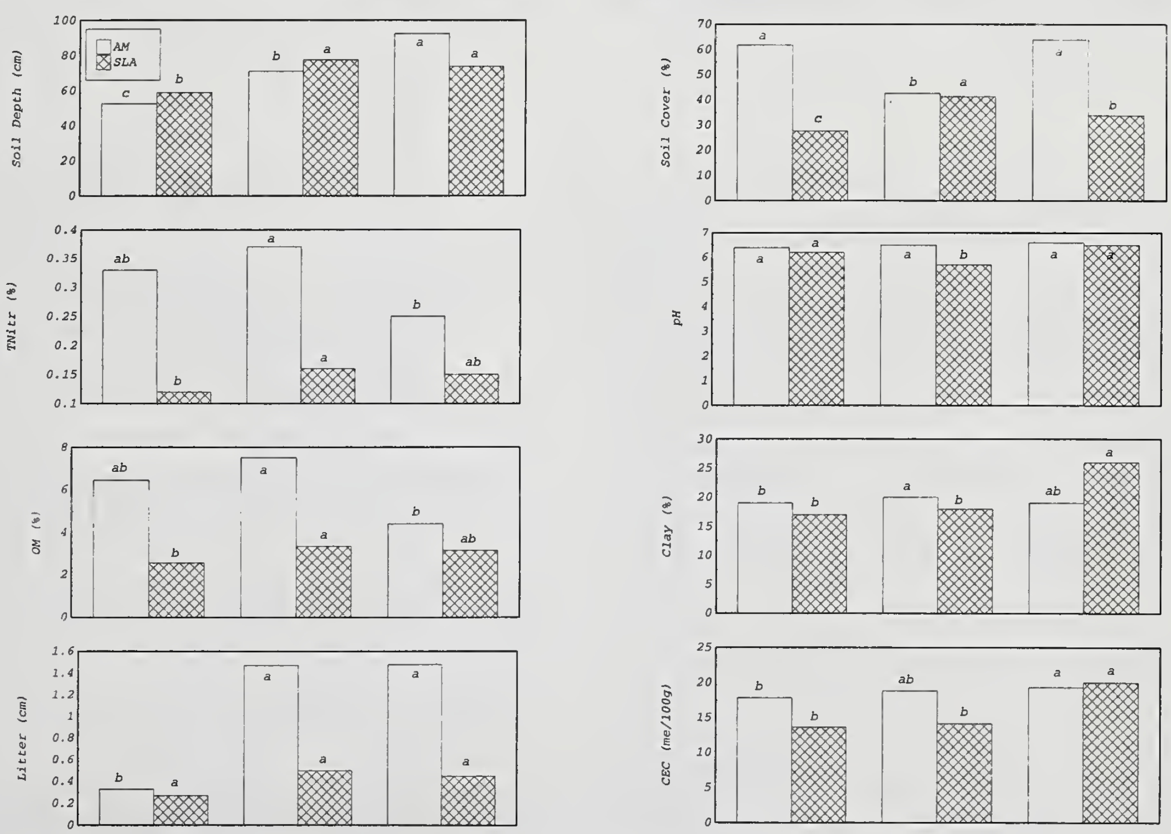


Figure 2. Effect of interaction between mountain range with forest community on morphological and chemical variables for Animas Mountains, New Mexico and Sierra los Ajos, Sonora. Bars with the same letter within a mountain range are not different (p >0.05).

found that the effects of repeated burning in ponderosa pine forests were a net increase in soluble N and increased rates of microbial mineralization stimulated by improved soil moisture and temperature. In the present case the SLA communities subjected to more frequent fires did not show increases in total nitrogen. This situation could be attributed to the fact that the influence of fire on nutrient availability may have taken place immediately or some months after the fire, and probably the favorable effect had already disappeared by the time the soils were sampled.

Clay content, cation exchange capacity (CEC) and pH were very similar for communities in the AM. In contrast, upper elevation communities in the SLA indicated higher values for clay content and CEC (Fig. 2, 3, and 4). pH was similar for PJ and Df/GO communities in the SLA.

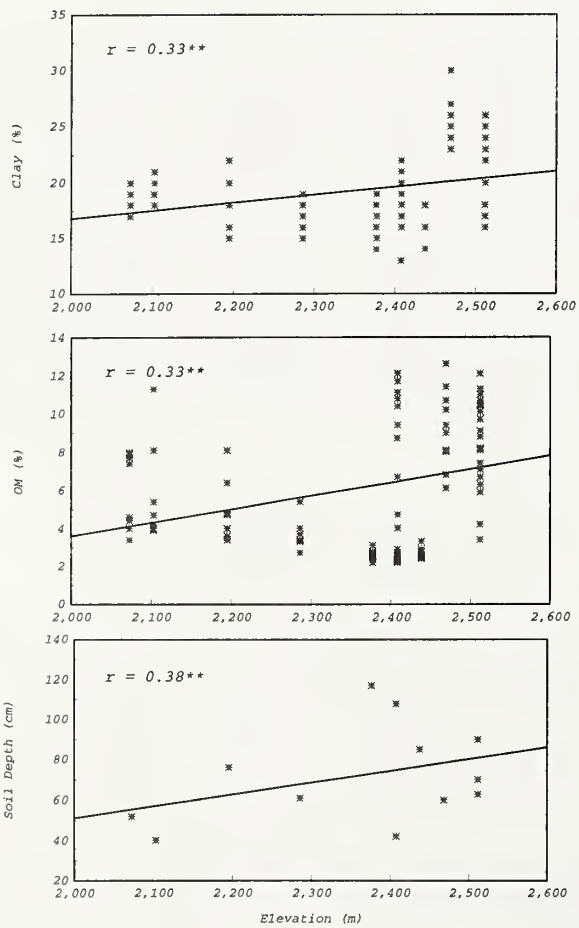
CEC is highly influenced by different factors such as clay content, type of clay, and organic matter content. Land-use history may indirectly influence CEC by affecting the amount of organic matter present. However, environmental factors probably have a greater influence on CEC values through weathering and chemical reactions (Whittaker et al. 1968, Barton 1994).

Main Effect (Mountain Range)

Independently of forest community, rockoutcrop, cobble, and stone on soil surface were nonsignificant between mountain ranges. Electrical conductivity and most of soluble ions were statistically higher for the AM as compared to the SLA. Differences between mountain ranges with relation to climate, soil, and land-use histories may explain some of those differences.

Main Effect (Forest Community)

Independent of mountain range, average values for available phosphate, soluble ions (Ca, Mg, CO3, HCO3, and Cl), and exchangeable potassium were statistically similar from lower to upper elevation communities.



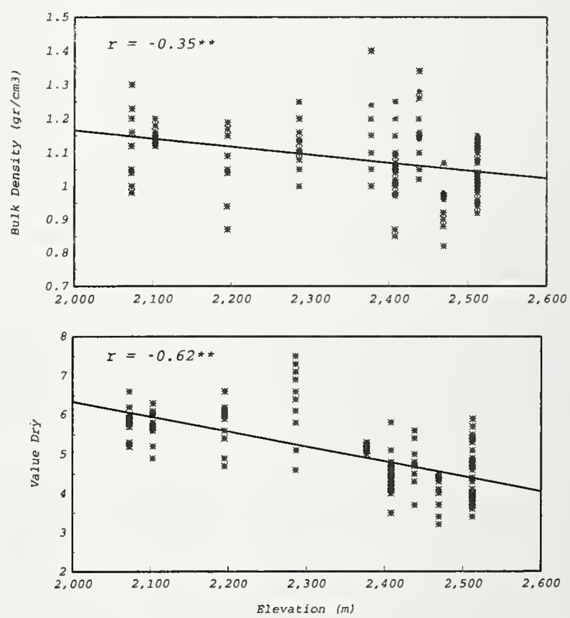


Figure 3. Pearson's Product Moment (r) relationship between elevation and edaphic variables for Animas Mountains, New mexico. Values are significant (p <0.01) when followed by **

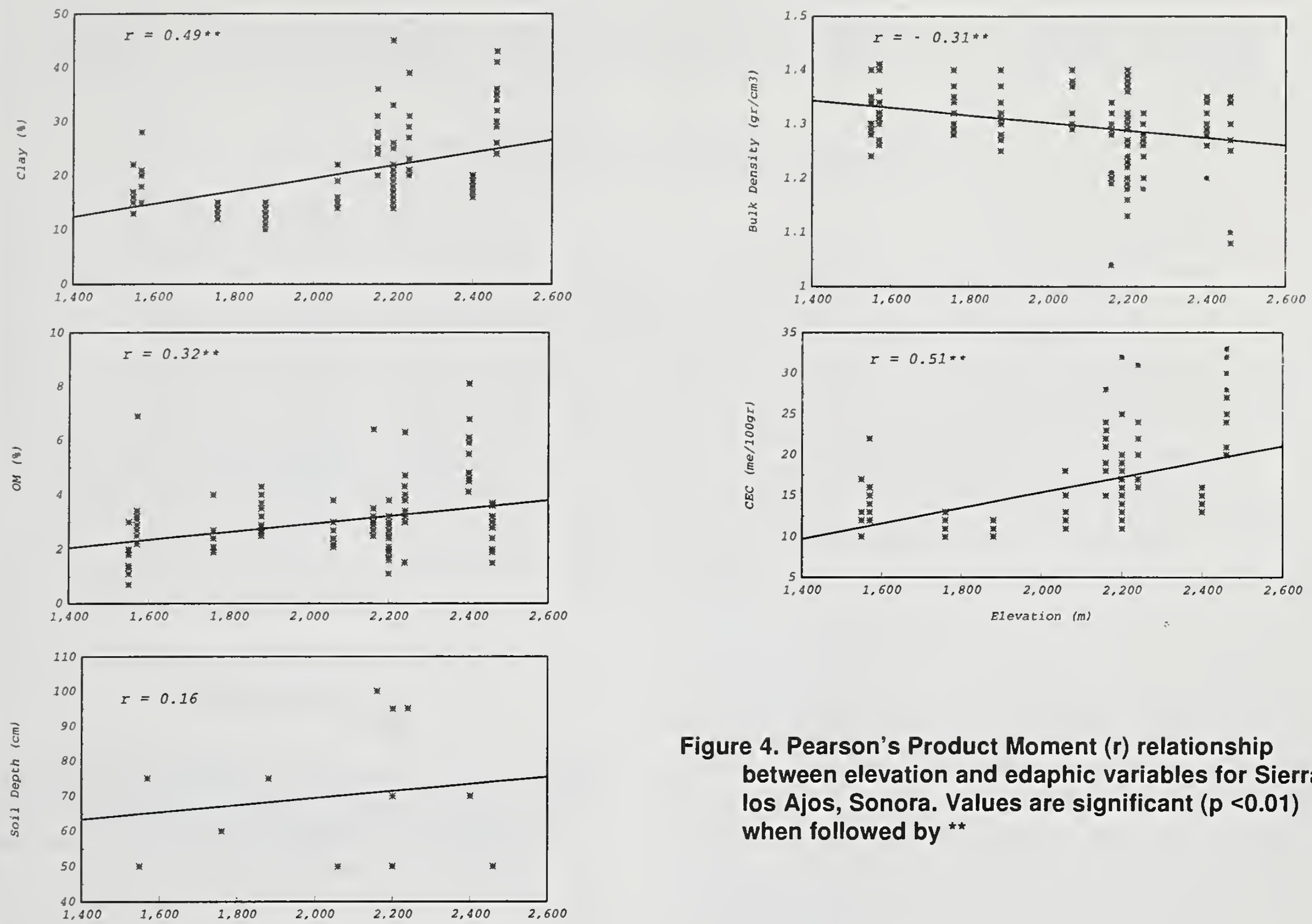


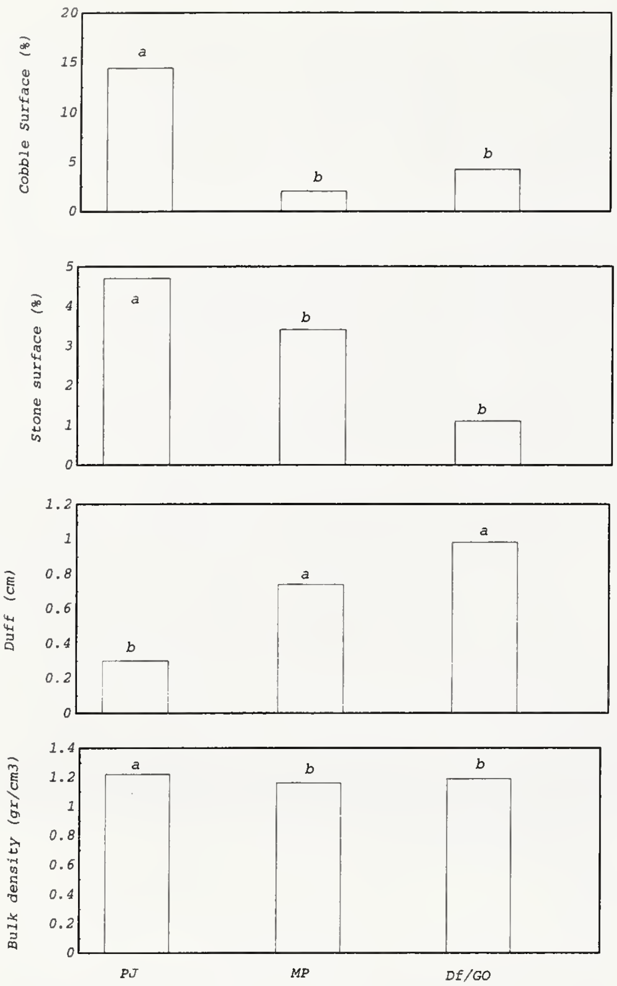
Figure 4. Pearson's Product Moment (r) relationship between elevation and edaphic variables for Sierra los Ajos, Sonora. Values are significant (p <0.01) when followed by **

Average bulk density was greater for PJ communities and lower but not statistically different for MP and Df/GO communities. This result is highly related to organic matter and clay content which were generally lower in PJ communities (Fig. 5).

Average duff depth increased from lower to upper elevation communities. This difference could be attributed to the presence of environmental conditions favoring increases in biomass production from lower to upper elevation communities, and thereby affecting rates of litter deposition and decomposition.

Rock fractions on soil surface (e.g., stones and cobbles) decreased from lower to upper elevation communities. The distribution of rock fractions in a given mountain range is also related to the underlaying geological formation, geological uplift, and erosion processes (Boyles and Tajchman 1984). We consider that climate (precipitation and temperature) and vegetation (due to root expansion and releases of carbon dioxide and other metabolic components product of physiological activity) at higher elevations enhanced rates of weathering and chemical processes, which reduced the amount of soil material greater than 2 millimeters in diameter. Comparatively, this weathering is reduced at lower elevation where chemical processes are limited by moisture availability.

Some of the chemical and morphological characteristics in communities of the AM and the SLA were significantly affected by an interaction of mountain range with forest community. However, the effect of land-use history, especially fire, did not always enhance nutrient availability. Different studies have indicated that fires may have a substantial impact on nutrients in soil-plant systems. However, the magnitude and direction of changes produced in the nutrient regime by fires, as well as the rate of recovery vary widely depending on burning conditions and environmental characteristics of the sites (DeBano and Conrad 1978). On the other hand, soil temperatures generated by wild fires or prescribed burns usually do not reach the magnitude necessary to influence soil properties below about 10 centimeters in the soil. Sampling in this study included the upper 25 centimeters of soil.



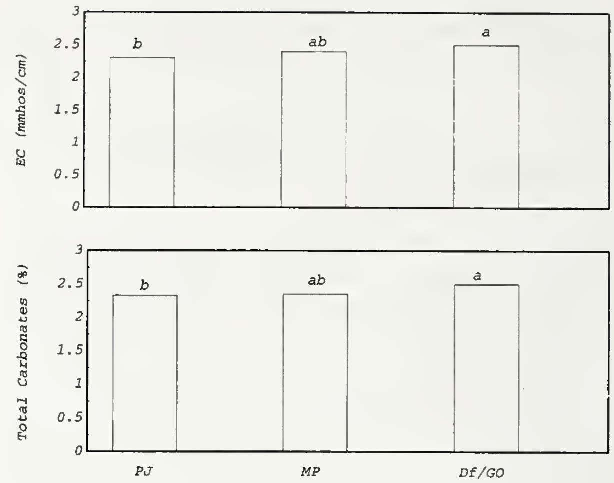


Figure 5. Effect of forest community (main effect) on morphological and chemical variables for Animas Mountains, New Mexico and Sierra los Ajos, Sonora. Bars with the same letter are not different (p >0.05).

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Peak Fire of 1988: Its Effect on Madrean Oak Trees

Peter F. Ffolliott[1] and Duane A. Bennett[2]

Abstract.—The human-caused but unintentional Peak Fire burned more than 12,000 acres in northern Sonora, Mexico, and southeastern Arizona from June 10 to June 17, 1988. The relation of the fire's severity to the occurrence of fire-damaged oak trees, or lack thereof, and the subsequent recovery of fire-damaged oak in 1996 indicate that the character of the oak resource in the burned area has been altered by the killing of trees, damaging of trees, and increasing the incidence of sprouting in instances where trees were shoot-killed. While sites burned with low fire severity might approach pre-fire stocking conditions sometime in the future, the high occurrence of root-killed oak on sites burned at higher fire severity suggests that the low stocking conditions observed eight years after the fire will probably persist on these sites.

INTRODUCTION

The human-caused but unintentional Peak Fire burned more than 12,000 acres in northern Sonora, Mexico, and southeastern Arizona from June 10 to June 17, 1988. The effect of this fire on Madrean oak trees, mostly Emory oak (*Quercus emoryi*) with intermingling Arizona white oak (*Q. arizonica*), has been studied since shortly after the fire was contained. The relation of the fire's severity to the occurrence of fire-damaged oak trees, or lack thereof, and the subsequent recovery of fire-damaged oak trees are being evaluated as part of this study. This paper presents the results of this evaluation in 1996, eight years after the fire.

THE FIRE

The Peak Fire, one of the largest fires to occur in the northern Madrean Province in recent years, began on the morning of June 10, in Sonora, Mexico, about one mile from the international boundary. Smoke from the initially small fire was reported by officials of the Coronado National Memorial. However, an attack by fire-fighting crews was not possible, because the fire's origin was Mexico. Early that afternoon, the fire crossed into Arizona with a 6-mile-wide front, at which time efforts were begun to protect residential and other cultural structures in the path of the fire. Shortly thereafter, the fire swept over Coronado Peak into Montezuma Canyon. A lack of suppression-action on the Mexican side, due to limited personnel and access to the remote area of the fire, allowed the fire to continually out-flank all of the initial controlling efforts.

The fire passed through the Memorial, over the next ridge and into Ash Canyon by late afternoon of June 11, burning more than 4,000 acres. The fire continued to actively burn in the Coronado National Forest for four days, ultimately impacting more than 12,000 acres in Sonora and Arizona. During the fire, an agreement was signed by officials in Washington, D.C., and Mexico City to allow fire fighters from the United States to enter Mexico to extinguish the remaining fire in Sonora. This agreement remains in effect.

The Peak Fire was officially designated controlled on June 17, 1988, with a suppression cost of $1,500,000. It was estimated by a post-fire evaluation that 30 percent of the burned area had experienced a low fire intensity, 35 percent a medium fire intensity, and 35 percent a high fire intensity.

EFFECT OF THE FIRE ON MADREAN OAK TREES

The occurrence of fire-damaged Madrean oak trees, and oak trees with no visible damage in 1996 (the

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more recent evaluation) in relation to sites that had experienced low fire severity, medium fire severity, and high fire severity is presented in this paper. Stocking conditions on these sites provide an indication of the long-term recovery of the trees after the fire.

Study Methods

The site representing low fire intensity was located on the Coronado National Forest, below the Montezuma Pass Overlook at about 6,500 feet in elevation, on largely southwestwardly slopes ranging from 25 to 40 percent. An area indicative of a medium fire severity was located on the Coronado National Memorial at 5,800 feet in elevation, on eastwardly slopes of 15 to 25 percent. The site characterizing high fire severity extended along the ridgeline from the Overlook to Coronado Peak; elevations on this site ranged from 6,650 to 6,800 feet, on both eastwardly and westwardly slopes of 10 to 25 percent.

Woody plants found on the three sites in addition to Madrean oaks included alligator juniper (*Juniper deppeana*) trees and manzanita (*Arctostaphylos patula*) shrubs. Many of the larger alligator juniper were severely damaged by the burn and, as a consequence, subsequently died. Manzanita has sprouted vigorously on all sites since the fire. An herbaceous understory of mostly native perennials also grows on the sites.

Oak trees were tallied on 30 1/20-acre sample plots located at 100-foot intervals along a series of temporarily-located transects that traversed each study site. Species recognition was not possible in many instances because of burning damage, and subsequent insect attacks and decomposition; therefore, differentiation between the oak species was not made. Trees tallied on the plots were classified with respect to fire damage as follows:

- No visible damage;
- Scorched crowns, in which case the occurrence or absence of sprouting was noted;
- Shoot-killed (sprouting);
- Root-killed (no sprouting).

Results and Discussion

A total of 271 oak trees was tallied on the sample plots representing low fire severity, 288 oak trees on the sample plots experiencing medium fire severity, and 171 oak trees on the sample plots characterized by high fire severity. The low tree tally on the site burned at a high fire severity was attributed to the complete consumption of many trees by the fire.

The percent of oak trees in the respective fire-damage classes in relation to the fire severity of the three sites studied is shown in figure 1. One-half of the trees tallied on the site with low fire severity exhibited no visible damage. However, the percent of oak with no visible damage significantly decreased as fire severity increased.

A greater percent of oak with scorched crowns has been observed on the site burned with medium fire severity than on the site of low fire severity. Most of the trees in this fire-damage class on both of the sites survived the fire, however. There were only a few trees with scorched crowns on the site representing high fire severity, where either the trees were consumed at the time of burning, or the fire killed the shoots or roots.

Shoot-killed oak ranged from 15 to almost 25 percent of the trees tallied on the three sites. Vigorous sprouting from basal buds below the ground-line has occurred around most of these trees since the fire. While this sample was small, the frequent occurrence of sprouting by shoot-killed Madrean oak after fire is consistent with that observed by Caprio and Zwolinski (1992, 1995).

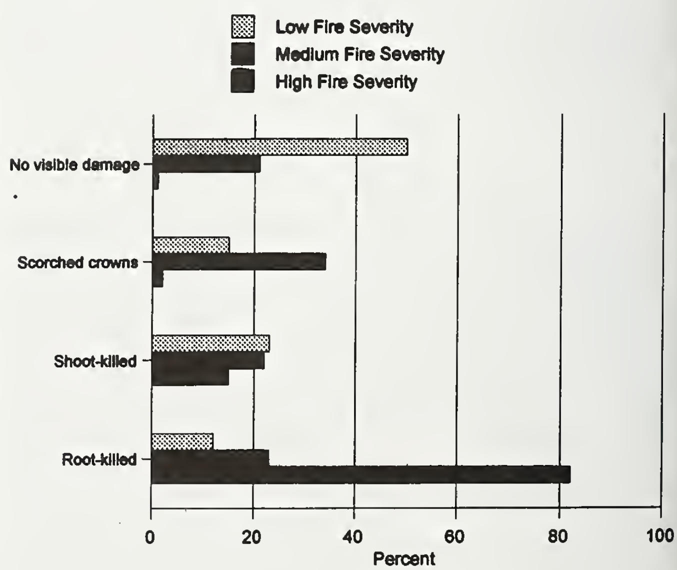

Figure 1. Percent of oak trees in fire-damage classes in relation to estimated fire severity.

Over 80 percent of the trees on the site of high fire severity were root-killed by the burn. Most of the other oaks tallied where shoot-killed. While all of the sample plots on this site were stocked with oak before the burn, only 16 plots (about 53 percent) were stocked with trees, sprouts, or both in 1996. It appears unlikely, therefore, that stocking of oak will return to pre-fire conditions on this site, largely because of the infrequency and unreliability of natural regeneration of Madrean oak species from seed (Pase 1969, Borelli et al. 1994).

MANAGEMENT IMPLICATIONS

Fire suppression in the past has resulted, in general, in a decrease in the frequency of fire in the region of the Peak Fire. As fire frequency decreases, episodic fires that do occur are likely to increase in severity; a consequence of this could be more fires that replace tree stands and, depending on the fire severity, increase the dominance of sprouting shrub species such as manzanita. A similar hypothesis was offered by Caprio (1994) in his study of fire effects on Madrean oak species in the Santa Catalina Mountains.

There is no doubt that the Peak Fire has altered the character of the oak resource in the burned area by killing trees, damaging trees, and increasing the incidence of sprouting (in instances where trees were shoot-killed). While sites burned with low fire severity might approach pre-fire stocking conditions sometime in the future, the high occurrence of root-killed oak on sites burned at higher fire severity suggests that the low stocking conditions observed eight years after the fire will probably persist on these sites. The low probability of obtaining successful natural regeneration of Madrean oak from seed also indicates the unlikelihood of re-stocking these latter sites with oak species except, perhaps, through sprouting.

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Fire History and the Possible Role of Apache-Set Fires in the Chiricahua Mountains of Southeastern Arizona

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Abstract.—Fire history was reconstructed for the Rustler Park area of the Chiricahua Mountains and compared with the historical and documentary record of Apache presence to interpret possible associations between the Apaches and fire occurrence. Dendrochronological techniques were used to crossdate and analyze samples from 63 fire-scarred trees, resulting in a tree-ring reconstruction of fire history that extended from 1644 to 1995. The fire chronology exhibited unusually high fire frequency relative to most other Southwestern fire chronologies (approximately one fire every three years between 1700 and 1900). A greater proportion of dormant season scars (late winter or spring fires) than observed elsewhere in the Southwest may indicate a greater occurrence of human-set fires. Specific key events in borderlands history were also concurrent with temporal changes in the Chiricahua fire chronology. For example, fire occurrence increased between 1760 and 1786 when Apaches waged an aggressive war against the Spanish, but decreased following 1786 when peace was established. While the hypothesis of important Apache alteration of fire regimes in the Rustler Park area was supported by concurrence with the documentary record and temporal patterns of fire occurrence, we could not conclusively distinguish the Apache influence from other factors regulating fire regimes, especially climate.

INTRODUCTION

During recent decades fire has assumed a more prominent role in land management, and is now acknowledged to be an essential ecological process in grassland and forest ecosystems. An understanding of the frequency, extent, seasonality, and severity of past wildfires is therefore required to help develop sound fire management policy. Developing knowledge of past fire regimes, however, is complicated by recent (post-1880) anthropogenic disturbances, such as logging, livestock grazing, mining, urban development, and fire suppression (Bahre 1991, 1995), that have disrupted fire as an ecological process. The application of dendrochronological (i.e., tree-ring dating) techniques to fire history studies allows researchers to evaluate the historical patterns of fires across both time and space prior to Euro-American disturbances (Swetnam and Baisan, in press). Such information identifies the historical range of variability of past fire regimes, providing a perspective on how ecosystems operated in the past, and the extent to which they have changed during the 20th century. Knowledge and understanding of these historical patterns provide fundamental evidence for development of ecologically informed land management plans (Allen 1994, Kaufmann et al. 1994, Swetnam and Baisan, this volume).

In southeastern Arizona, the Chiricahua Apaches were the dominant force in shaping local history from the approximate time of their arrival into southeastern Arizona (perhaps as late as the 1600s), until the final surrender of Geronimo in 1886 (Opler 1983, Worcester 1979). The Apache economy was based heavily on raiding and warfare supplemented by hunting and gathering, making it difficult for both Spanish and American forces to secure the South-

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west for settlement and exploitation. In 1748, war officially was declared against the Apaches by the Spanish; however, a state of war had been in effect for many previous decades with Apache raids throughout southern Arizona, New Mexico, and northern Sonora (Wilson 1995). Between 1760 and 1886, the Apaches were the only Native American group that challenged Spanish, Mexican, and American control of southeastern Arizona, traveling to and from northern Sonora via the San Pedro, Sulphur Spring, and San Simon Valleys to plunder rancherias throughout the borderlands area.

Native Americans of the Southwest allegedly used fire for various reasons: for driving game, to increase forage for livestock and large game, for direct and indirect warfare tactics, and for creating travel corridors (Dobyns 1981, Pyne 1982). Because the nomadic Apaches lived and traveled extensively throughout southeastern Arizona, it is possible that fire regimes were altered by their use of fire in some areas during certain periods. Newspaper reports from the late 1800s contain several allegations of Apaches setting fires in the mountains around Tucson, Arizona (Bahre 1985). However, early accounts of wildfires set by Apaches should be interpreted cautiously because:

1. Only a few eye-witness accounts of Apaches actually setting fires are known,
2. Most fires attributed to Apaches occurred during the time of year when lightning fires were most common, and
3. Early Euro-American settlers expressed considerable anti-Apache sentiment, and may have been prone to attribute any fire to the Apaches (Baisan 1990).

Swetnam and Baisan (in press) have argued that, while Native Americans probably altered fire regimes during certain periods and in certain places in the Southwest, lightning patterns, fuel moisture, and fuel production largely controlled fire regimes in most Southwestern locations. Thus, the debate is not whether Native Americans intentionally burned landscapes or not, but rather where and when human-set fires influenced Southwestern fire regimes.

If Native American groups altered past fire regimes, then distinct patterns of fire occurrence indicating such use may be apparent in certain fire chronologies as:

1. Periods of increased fire frequency above the frequency expected from lightning fires alone,
2. Temporal changes in fire regimes that correspond with documented events in Apache history, or
3. Some other change in the historical fire regime pattern (e.g., a shift in the seasonality of past fires that is not explained by concurrent changes in climate).

The purpose of this study was to examine the relationship between wildfires and Apaches by developing a tree-ring based fire history for the Rustler Park area of the Chiricahua Mountains, and comparing this fire chronology to the historic record of Apache presence to determine their possible influence on fire occurrence in the Chiricahua Mountains.

SITE DESCRIPTION

The study site was located in the high-elevation mixed-conifer forests along the crest of the Chiricahua Mountains (Fig. 1) in and around the Rustler Park area. This area was selected because the high-elevation meadows of Rustler Park are in close proximity to travel routes over the mountains, and to areas used by the Chiricahua Apaches, and therefore might reasonably be expected to contain a record of their influence on fire regimes. The most common habitat type is *Pseudotsuga menziesii-Pinus strobiformis/*

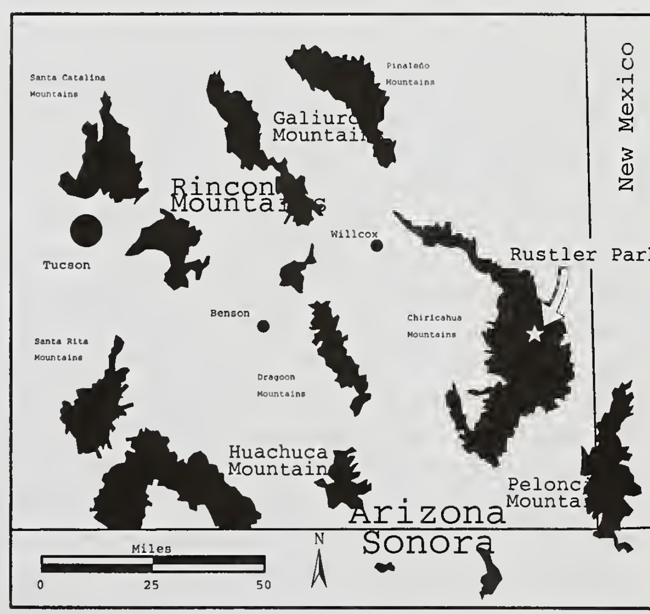

Figure 1. Location of the study site in the Chiricahua Mountains. Shaded areas are major forested and woodland areas of southeastern Arizona.

Muhlenbergia virescens (Moir and Ludwig 1979), dominated by Douglas-fir (*Pseudotsuga menziesii*) and southwestern white pine (*Pinus strobiformis*). Common associates include ponderosa (or Arizona) pine (*Pinus ponderosa*), white fir (*Abies concolor*), and quaking aspen (*Populus tremuloides*). Common understory plants include screwleaf muhly (*Muhlenbergia virescens*), creeping barberry (*Berberis repens*), western bracken fern (*Pteridium aquilinum*), false lupine (*Thermopsis pinetorum*), and Pringle's needle grass (*Stipa pringlei*) (Moir and Ludwig 1979, Sawyer and Kinraide 1980). Extensive and "abusive" logging occurred in these high elevation forests beginning in the 1870s (Bahre 1995), but subsided with the establishment of the Crook National Forest (later incorporated into the Coronado National Forest) in 1908. Livestock grazing, a major industry in the nearby Sulphur Spring Valley beginning in the late 1870s (Bailey 1994), occurred throughout the mountain range (Bahre 1995), and may have considerably impacted the high elevation forests as well.

METHODS

We used a chain saw to cut cross-sections from 63 fire-scarred southwestern white pine and ponderosa pine logs, snags, stumps, and remnant pieces of wood along a two-mile north-south gradient, beginning at the trail head to Buena Vista Peak, through Rustler Park, and culminating in the Long Park area. Smaller cross-sections were cut from living trees to obtain a record of 20th century fires (Arno and Sneck 1977). In the laboratory, all tree-ring samples were sanded, then crossdated using skeleton plots developed from nearby tree-ring chronologies in the Chiricahua Mountains (Pinery Canyon and Flys Peak). The seasonal timing of past fires was inferred from the position of the fire scars within the annual rings (Baisan and Swetnam 1990). All information was entered in a computer database, then statistically and graphically analyzed using FHX2, software designed specifically for analyzing fire history from tree rings (Grissino-Mayer and others 1994, Grissino-Mayer 1995). Statistical analyses incorporated the Weibull distribution fit to the fire interval data to provide more robust statistical measures of fire regime characteristics (Grissino-Mayer 1995). Finally, we compared the composite fire chronology to documentary evidence of Apache presence in southeastern Arizona to identify coincidences (or lack thereof) in Apache historical events that would, by inference, suggest the possible influence of Apache-set fires in altering fire regimes in the study area.

RESULTS

The master fire chronology revealed that wildfires occurred frequently in the Chiricahua Mountains prior to 1892 (Fig. 2). Between 1700 and 1892, the period when sample depth was considered adequate for statistical analyses (i.e., a minimum of five trees recorded fires), fires occurred approximately once every three years based on the mean, median, and Weibull median probability fire intervals (Table 1). The shortest interval between fire years was one year, which occurred several times during the 200 year period (e.g. 1759-1760, 1765-1766, and 1862-1863). The most remarkable sequence of one year intervals occurred during the years 1770 (recorded

Table 1—Descriptive statistics for fire intervals, Chiricahua Mountains, 1700 - 1900. All values are in years.

	Statistic
Mean Fire Interval	2.91
Median Fire Interval:	3.00
Weibull Median Probability Interval[a]:	2.71
Fire Frequency Probability per Year[b]:	0.37
Standard Deviation:	2.09
Coefficient of Variation:	0.72
Weibull Shape Parameterc:	2.30
Skewness:	3.88
Kurtosis:	21.45
Minimum Fire Interval:	1.00
Maximum Fire Interval:	16.00
95% Exceedance Intervald:	0.87
5% Exceedance Intervale:	5.12
Maximum Hazard Intervalf:	4.00

[a] *Median interval based on fitting a Weibull distribution to the fire interval data.*

[b] *Inverse of the WMPI.*

[c] *Shape parameter estimated for the Weibull distribution fit to the actual data.*

[d] *Interval exceeded by 95% of all other intervals based on the Weibull distribution.*

[e] *Interval exceeded by 5% of all other intervals.*

[f] *Maximum interval at the 100% hazard rate based on the Weibull distribution.*

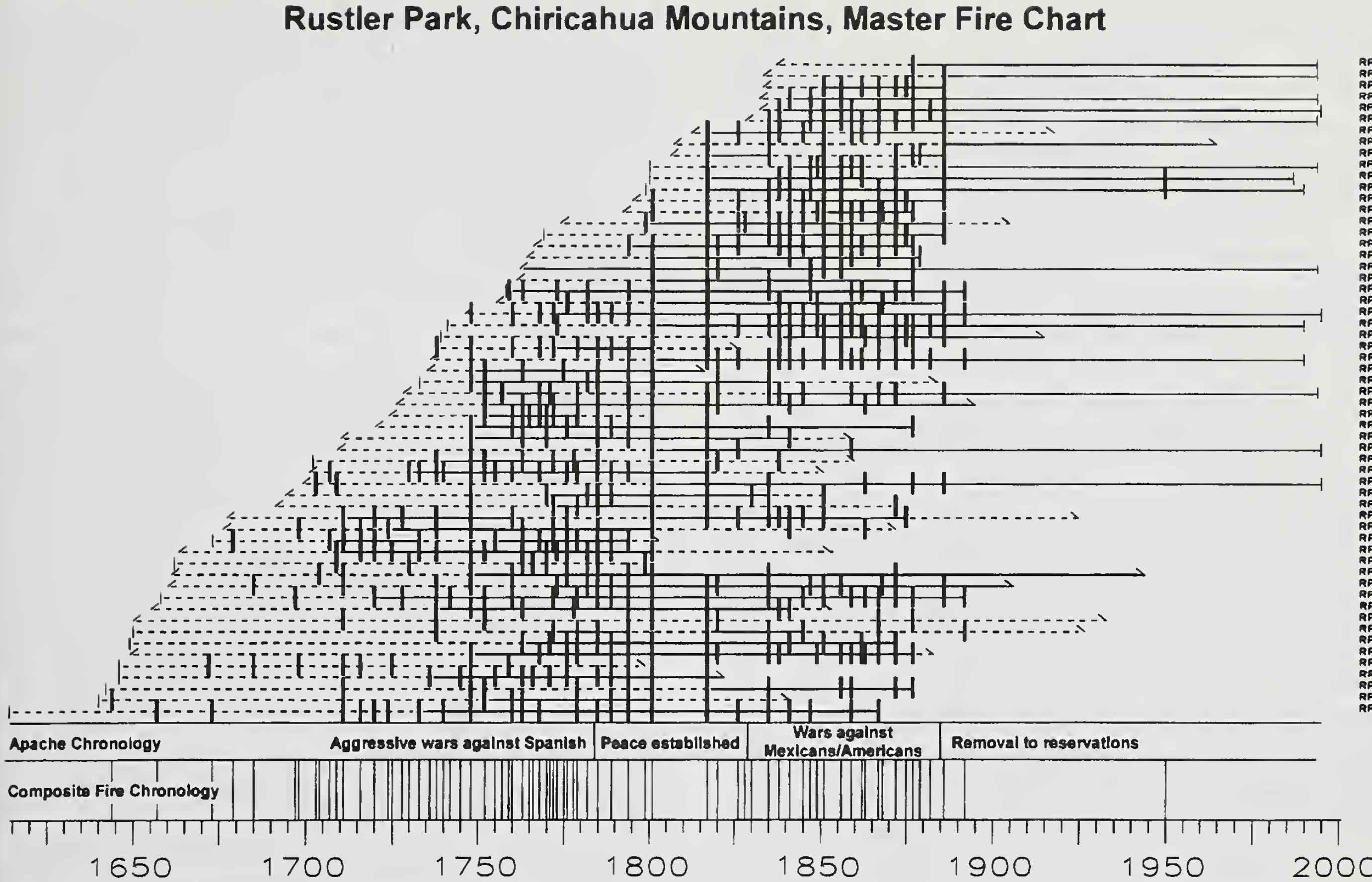

Figure 2. Master fire chronology for the study site. Each horizontal line represents information from one tree, while small vertical bars represent dated fire events (i.e. fire scars or other fire related injuries). Note the long fire-free interval between 1801 and 1817, and the near absence of fires after 1890.

by five trees), 1771 (three trees), 1772 (10 trees), and 1773 (four trees). Such long runs of consecutive-year fires have only been reconstructed in southern Arizona and New Mexico at one other location - the Organ Mountains of south-central New Mexico where Apache-set fires were also hypothesized to have been especially important (Morino 1996).

After the fire in 1892, however, widespread fires virtually ceased, a common feature of nearly all fire-scar chronologies so far developed for the Southwest (Swetnam and Baisan, in press). This rather sudden decrease in fire frequency is attributed primarily to two human-related disturbances:

- Grazing by livestock (Allen 1994, Grissino-Mayer and others 1994, Savage and Swetnam 1990, Touchan and others 1995) which became a major industry of southeastern Arizona beginning in the 1880s (Allen 1989, Bahre 1991, Bailey 1994, Wilson 1995), and
- Fire suppression, the effects of which are most pronounced after World War II (Grissino-Mayer 1995, Pyne 1982, van Wagtendonk 1991), although some suppression efforts may have begun as early as the 1910s.

Many fires were widespread along the entire two mile gradient (e.g., the fires in 1748, 1801, 1817, 1851, and 1886). Synchroneity with past fires in Chiricahua National Monument 8.5 miles to the north (Swetnam and others 1989) and with fires in other lower-elevation locations in the Chiricahuas (Kaib and others, this volume) indicates that some widespread fires prior to 1890 were perhaps equal to or greater than the size of the Rattlesnake Fire of 1994 in the Chiricahua Mountains (ca. 27,500 acres).

The longest interval between fires was 16 years between 1801 and 1817 (Fig. 2). This long fire-free period was concurrent with similar unusually long fire-free periods in many other mountain areas of the

Southwest (Swetnam and Baisan, in press and this volume). Critical intervals, denoting unusually long fire-free periods, are provided by the 5% exceedance interval and the maximum hazard interval (Grissino-Mayer 1995), which indicate that fire-free periods of four to five years approached the maximum length the forests of the Chiricahua Mountains sustained in the presettlement era before burning was highly probable. Therefore, the unusual 16 year fire-free period far exceeded the maximum length expected based on the fire interval distribution.

The seasonality of past fires was strongly bimodal (Fig. 3), with peaks occurring in both the dormant season (i.e. in spring, prior to the onset of tree growth) and the middle portion of the growing season (i.e. late May to early June). This seasonal pattern is unlike that found for most other fire histories in the borderlands, which typically showed that fires occurred predominantly during the early and middle portions of the growing season prior to the onset of summer monsoonal rainfall (Swetnam and Baisan, this volume). For example, in the nearby Pinaleño Mountains to the north, over 70% of all fire scars were positioned in the early and middle portions of the

Figure 3. The seasonality of past fires in the Chiricahua Mountains based on the positions of fire scars within the annual rings. The distribution is strongly bimodal. Approximate periods associated with these seasons: Dormant—prior to mid-May; Early Earlywood—mid-May to early June; Middle Earlywood—late May to early July; Late Earlywood—mid-June to late July; Latewood—after July. Overlaps in these seasonal designations occur due to differential growth associated with year-to-year climate variations.

tree rings (Grissino-Mayer and others 1994). Analyses on temporal patterns of seasonality (Swetnam 1992; Grissino-Mayer 1995) for the Rustler Park fire chronology showed no period was dominated by one scar position over another. The distribution of scar positions within individual fire years (Baisan and Swetnam 1990) revealed presettlement fires may have burned throughout most of the growing season during some fire years. For example, the major fire year of 1817 at Rustler Park showed 25% of all scars were classified as dormant season, 18% as early growing season, 46% as middle growing season, and 11% as late growing season (n=28). Few fire years were dominated by only one scar position.

DISCUSSION

To investigate whether Apaches may have altered past fire regimes in the Chiricahua Mountains, we compared specific historical events in Apache history with the Chiricahua fire chronology (Table 2). Certain key periods in Apache history coincided well with specific changes in the fire chronology. For example, fire occurrence increased in the latter decades of the 17th century, a period when the Apaches resided in and around the Chiricahua Mountains, and made repeated raids into northern Mexico (Wilson 1995, Worcester 1979). However, the low fire frequency prior to 1690 may be an artifact of low sample depth rather than specific changes in Apache behavior. The first major widespread recorded fire occurred in 1748, the year in which the Spanish officially declared war on the Apaches. However, fires occurred in numerous mountain ranges throughout the Southwest during this extreme drought year (Swetnam 1990, Swetnam and Baisan, in press), indicating that regional climatic patterns were largely responsible for the widespread fires in this year.

Fire occurrence increased between 1760 and 1780 (Fig. 2), a period when the Apaches aggressively fought the Spanish (Wilson 1995). This period was also characterized by an unusual number of one-year fire intervals, lending support to the hypothesis for Apache alteration of fire regimes, because one-year intervals are uncommon in most Southwestern fire chronologies. Fire may have been used extensively by the Apaches as a means for warfare (e.g., intentional burning of vegetation to deter the enemy, to drive the enemy to prearranged areas, to deprive

Table 2—Chronology comparing major events in Apachean history (adapted from Wilson 1995) with major features of the fire history developed for this study.

Year	Events in Apachean History	Rustler Park Fire Chronology
1682	Apaches made first reported raid into Mexico.	Fire chronology shows increased numbers
1690s	Apaches are known to reside in the Chiricahua mountains.	of fires beginning late 1600s.
1748	Spanish viceroy approves a declaration of war against the Apaches.	First major widespread fire. Over 85% of trees scarred in study area.
1760	Apaches begin aggressive challenge of Spanish control of southeastern Arizona.	Fire frequency dramatically increases between 1760 and 1780.
1768	Apaches launch major offensive into Sonora.	
1786	Peace established with Apaches.	Wildfires decrease dramatically.
1831	Apaches resume raiding.	Wildfires resume in 1817, but long gap
1843	Apaches force abandonment of last Mexican ranches in northern Sonora.	until next major fire in 1835.
1861	War breaks out between the Apaches and the United States army.	Wildfires continue at short, fairly regular
1872	Cochise agrees to go onto a reservation, and the Apaches are now peaceful.	intervals.
1873	Heavy ranching develops in the Sulphur Springs Valley.	Wildfires continue at short, fairly regular
1878	Commercial lumbering and the earliest sawmill in the Chiricahua Mountains.	intervals.
1886	Geronimo surrenders; Indian wars over.	Last major wildfire in Chiricahuas.

forage for stock owned by their enemies, or to cover trails left by the Apaches, see Pyne 1982), thereby increasing the number of fire occurrences above that expected from lightning fires alone. After peace was established in 1786, fire occurrence dropped considerably, with no fires recorded between 1801 and 1817. However, peace with the Apaches continued until 1831 (Wilson 1995), but a widespread fire occurred in 1817, followed by other smaller fires in 1820, 1826, 1828, and 1830. Nonetheless, the decrease in fire occurrences between 1786 and 1831 is visually obvious (Fig. 2), was concurrent with the Apache peace, and was indeed followed by an increase in fire frequency after 1835.

However, fire occurrence after 1831 was not as common as fire occurrence prior to 1786. The mean fire interval during the period 1831-1900 (3.00 yrs, n=19) was significantly longer ($p < .05$) than the mean fire interval for the period 1700-1786 (2.28 yrs, n=36). Although this is a relatively small change, it is both graphically (Fig. 2) and statistically discernible. During the peace period between 1786 and 1831, the Spanish supplied rations to the Apaches, and Apache raids were less likely. Therefore, intentionally-set fires (perhaps for hunting and warfare practices) would have decreased. However, after the peace was broken and the Apaches once again turned to raiding and warfare, fire occurrences should have increased to pre-peace levels if the hypothesis of Apache-set fires is correct. It is possible that the long fire-free period between 1801 and 1817 may have caused fuels to become more homogeneous across the landscape, causing fewer, but perhaps more widespread fires. Similar temporal changes in fire regimes (from the late 1700s to early 1800s) have been observed in other locations in the Southwest, arguing in favor for a regional factor, such as climate. Grissino-Mayer (1995) attributed changes in fire regimes at El Malpais National Monument, New Mexico, to changes in long-term precipitation, especially in the summer monsoonal component. Morino (1996), however, attributed similar temporal changes in fire regimes for the Organ Mountains of southern New Mexico to Apache-set fires, but could not rule out changes in climate as a possible factor.

Fire occurrence terminated abruptly after the major fire in 1886, and another smaller fire in 1892. Interestingly, war between American forces and the Chiricahua Apaches also formally terminated in 1886 with the surrender of Geronimo. Throughout the Southwest, the period between 1870 and 1900 saw most Apachean groups deported and/or placed on reservations, concurrent with the decrease in fire occurrences throughout the Southwest. It is unlikely, however, that the post-1880 fire decline was directly due to the removal of Native Americans. For example, in the Sierra de los Ajos in northern Sonora, fire occurrence continued uninterrupted into the 20th century (Baisan and Swetnam 1995, and this volume), suggesting Apaches were not the cause of

wildfires. The post-1880 influx of Euro-Americans greatly disturbed the Southwestern landscape through livestock grazing, logging, mining, fuelwood gathering, urban development, and fire exclusion. These factors, particularly intensive livestock grazing, are widely cited as being primary causes for fire decline (Cooper 1961, Leopold 1924, Savage and Swetnam 1990, Swetnam 1990). In addition, a shift to wetter conditions around the turn of the century (Fritts 1991, Grissino-Mayer 1995) may also have contributed to a decrease in fire occurrences, although the shift occurred ca. 1910-1920, after the changes in fire frequency. Hence, the decline in surface fires in the Southwest may have been due to many factors, among them the physical removal of the Apaches from the Southwestern landscape.

The seasonality of past fires in the Chiricahua Mountains lends some support for the hypothesis of Apache-set fires. The high proportion of dormant season scars is unusual relative to fire histories developed for nearby sites, such as Rhyolite Canyon in Chiricahua National Monument (Swetnam and others 1989), the Camp Point and Peter's Flat sites in the Pinaleño Mountains (Grissino-Mayer and others 1994), and Mica Mountain in the Rincon Mountains (Baisan and Swetnam 1990). The Rustler Park fire scars showed a much higher proportion of dormant season scars than observed in these other sites. This pattern could be due to Apache-set fires during the fall and/or winter seasons, or in spring prior to the start of the growing season. Because most lightning ignitions occur predominantly in May through July and are less frequent in the dormant season, a larger proportion of fire scars in the dormant season argues for anthropogenic fires. Unfortunately, no accurate records of specific seasonal use of fire by the Apaches exists to support this hypothesis.

CONCLUSIONS

In summary, no firm conclusions can be drawn concerning the degree to which the Apaches influenced the fire regimes of the Chiricahua Mountains. It is highly probable that the Apaches exerted some influence, because they traveled the area extensively for 300+ years, and fire may have been a useful tool for specific purposes. Clear associations exist between the fire chronology and specific, major events in Apache history. During the wars with the Spanish, fire occurrence increased (with many one-year intervals), while fires decreased following a peace establishment in 1786. After the peace was broken in 1831, fires resumed, but occurred less frequently. The unusual, bimodal seasonality of past fires also argues for possible Apache influence on fire regimes, as does the termination of fires following the removal of the Apaches to reservations after 1886.

While some scholars believe that all Native American groups used fire extensively for many reasons (Pyne 1982, Sauer 1950), widespread intentional use of fire may have been incompatible with some aspects of the Apache economy, which was largely based on hunting, gathering, raiding, and warfare (Basso 1971, Opler 1983, Worcester 1979). Intentional use of fire would have exposed locations of Apache encampments, which relied heavily on stealth and secrecy, to enemy factions over broad areas. Intentional burning might also have been detrimental to many of the major food plants upon which the Apaches subsisted (e.g. agaves), but may also have increased forage for livestock taken in raids (Kaib and others 1996). In addition, Apache clans in the southern portion of the Southwest were mostly nomadic, occupying the valleys between mountain ranges. Temporary residences may have been established at higher elevations for hunting, gathering, and religious purposes, but long-term use of the higher elevations was not practiced. Finally, separation of Apache camps was needed to make available the limited resources in the semiarid deserts of the Southwest borderlands. Careless use of fire by one group could have been detrimental to adjacent camps (and therefore to relatives).

Additional research is needed to:

- Gather specific information about the extent to which the Apaches used fire, and
- Determine where and when Apache fire practices had significant impacts on fire regimes and, therefore, ecosystems of the borderlands region.

First, a more thorough search for and review of the available literature on the Apaches should be conducted to document traditional use of fire, including specific periods of use, locations, intent, and seasonal use of fire. This information should be compared with previous citations of use of fire by the Apaches. Second, reconstructions of fire history from fire-scar records should be obtained from specific areas fre-

quently used by the Apaches (e.g., known travel routes in southeastern and east-central Arizona) and compared to fire chronologies developed for areas unlikely to have been frequently used (e.g., see Barrett and Arno 1982). Support for the Apache-set fire hypothesis would be gained if the fire-scar records consistently showed different fire regimes in areas of frequent Apache use. Third, more detailed analyses concerning the wildfire-climate relationship should be conducted. Because climate and fire occurrence are coupled across both time and space, specific periods showing a weak or non-existent relationship would argue in favor of Apache alteration of fire regimes. For example, Swetnam and others (1990) showed no relationship existed between fire and climate during the period 1800 to 1900 in the lower portion of Rhyolite Canyon in Chiricahua National Monument, and hypothesized that Apache-set fires could have disrupted the fire-climate relationship.

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Postfire Saguaro Injury in Arizona's Sonoran Desert

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Abstract.—In May 1993, arson wildfires burned along Hwy 87 in saguaro-shrub vegetation on the Mesa Ranger District, Tonto National Forest, Arizona. Preliminary findings on the extent of saguaro injury caused by these wildfires are presented here.Height class distribution was similar for saguaros from unburned and burned areas. Saguaro mortality was about 2 percent on unburned sites compared to an initial 19 percent on burned sites. Over 90 percent of saguaros exhibited fire injury, and more than 60 percent were girdled. Long-term mortality may increase to over 80 percent of the saguaros on burned sites. Six woody small-tree or large-shrub species composed 88 percent of nearest neighbors associated with saguaro. An evaluation of the fuel potential of these nearest neighbors is needed.

INTRODUCTION

Wildfire frequency and acreage burned have increased over the past 40 years in upland Sonoran Desert communities (Rogers 1986, Schmid and Rogers 1988, Narog et al.1995). Superior giant saguaros, *Carnegiea gigantea* (Engelm.) Britt. & Rose, are being degraded or lost by this apparent change in fire occurrence (Rogers 1985, Wilson et al.1995a,b). Cave and Patten (1984), McLaughlin and Bowers (1982), and Thomas (1991) document similar effects of fire on desert vegetation. Although associated woody plants (e.g., foothill paloverde, *Cercidium microphyllum* (Torr.) Rose & Johnst.) are considered nurse plants by some (e.g., McAuliffe 1984, Turner et al. 1995), they may contribute to saguaro injury or death by providing fuel for fire (Wilson et al.1995a).

Freezing temperatures delimit the range of saguaro, and may unpredictably damage large numbers of them in a short period of time (Steenbergh and Lowe 1976, 1983). Fire may also destroy large numbers of saguaros (Cave and Patten 1984) when thousands of hectares of desert are burned (Narog et al.1995).

Programs addressing multiple use concerns and fire management in these desert ecosystems are difficult to develop (Wright 1988). In response to the need for an evaluation of wildfire effects on saguaro and its associated vegetation, a cooperative effort between the Tonto National Forest and the Pacific Southwest Research Station was initiated. In January of 1994, we began a project to study fire effects on saguaro-shrub vegetation, and investigate fire management options (Narog et al.1995). This paper compares preliminary findings on saguaro injury and mortality at unburned sites with fire-caused injury and mortality at burned sites on the Tonto National Forest, Arizona.

METHODS

Study Area

On May 4, 1993, arson fires were set between the Vista View Desert Observation Point along Hwy. 87 and Bush Hwy., Arizona. The extent of these fires and existence of adjacent unburned saguaro-shrub vegetation lead to the selection of this area for our permanent plots.

Study plots were located at an elevation of about 800 m, within a 25-square-km area known as "The Rolls" (R8E: T3-Sec. 2, 4, 5, and T4-Sec. 26, 27, 35 of the Four Peaks Quad), Mesa Ranger District, Tonto National Forest, Arizona. The heavily vegetated unburned areas have no record of recent fires. This open

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rangeland area is subject to heavy recreational use and intermittent livestock grazing.

Experimental Design

Sampling methods (Cox 1990) were modified to fit the characteristics of this saguaro-shrub vegetation. To minimize geologic, elevational, and vegetational gradients, plots were placed in and around the Vista View Burn (Narog et al. 1995, Wilson et al. 1995a) along similar aspect and slope contours. We used plotless point-quarter and nearest neighbor techniques to collect data from five point-quarter transects associated with five of our nine[1]-hectare plots: two in unburned and three in adjacent burned areas.

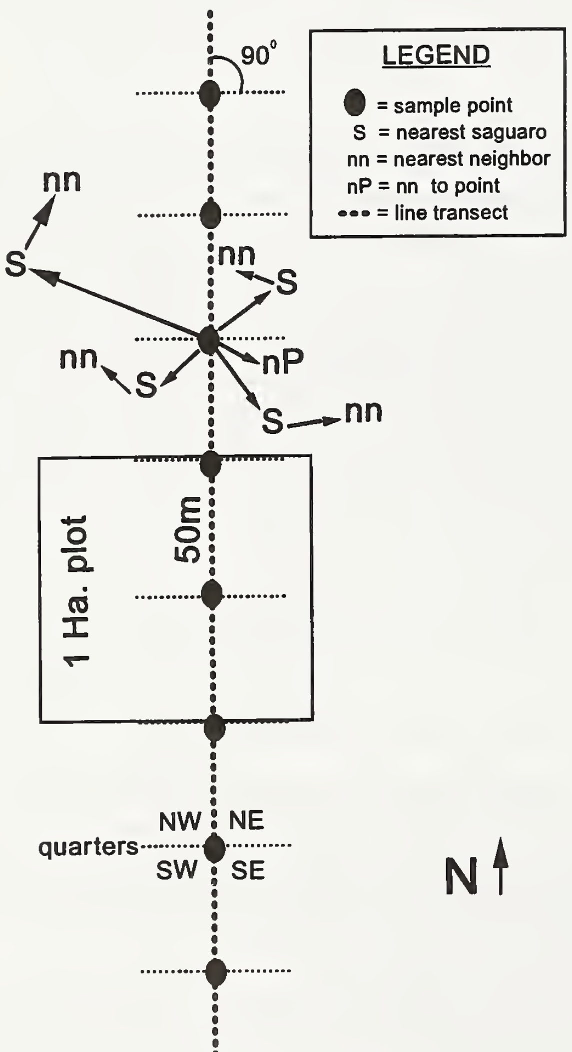

Figure 1. Point-quarter transect and nearest neighbor techniques used to locate saguaros and nearest neighbor plants on unburned and burned sites, Tonto National Forest, Arizona.

Transects started at a randomly selected point along the northern perimeter of each plot and extended 150 m to the north, south through the plot, and 100 m south of the plot (fig. 1). At each of eight points (50 m apart), data were taken for: 1) the nearest saguaro (within 200 m) and its nearest neighbor shrub in each of four quarters, and 2) the shrub nearest each point. When a saguaro occurred as the nearest saguaro to two adjacent transect points, the overlap was recorded and duplicate data were not taken.

Fire-caused saguaro injury was measured by: 1) degree of stem circumference affected, and scar height and aspect, and 2) whether the saguaro was alive or dead (See Analysis). Each saguaro sampled was photographed.

Analysis

Relative percent frequency ((saguaro parameter frequency / total frequency) X 100 = relative percent) was used to plot saguaro height data from unburned and burned sites, and to correlate fire scar data with aspect. We used Quatro Pro for windows statistics and t-test programs to compare height data. Saguaros from burned sites were grouped into four injury categories:

1. Dead—burned with no apparent signs of live green tissue or recently fallen,
2. Girdled—living, but burned 360 degrees around its circumference,
3. Scarred—not girdled but with at least some apparent fire damage, and
4. No scars—no discernible fire-caused injury.

Saguaro heights were sorted into 0.5 m height class intervals. Age estimates are based on height-to-age correlations for Saguaros from Tucson populations (Steenbergh and Lowe 1983).

RESULTS

Saguaro

A total of 137 saguaros were evaluated along the five transects. Individual overlap accounted for a loss of 22 of the 160 possible saguaros: 11 in unburned and 11 in burned areas. No saguaro occurred within 200 m in one quarter on a burned site.

Average saguaro heights from unburned (4.39±2.1 m) and burned (4.73±2.6 m) samples were similar (t = 0.5>P>0.1). Based on height-to-age correlations, the ages of the 137 saguaros are roughly estimated to range from about 20 years to 200 years. Of the 137 saguaros, individuals are noticeably absent in the 10 m to 11.5 m height classes, possibly representing a 50-60 year time period (fig. 2). The shortest saguaro sampled was 0.22 m. Seedlings were not observed in either unburned or burned areas.

On burned sites, frequency of fire-caused injury and mortality (fig. 3) is relatively even across saguaro height classes. Generally, fire damage was most severe at their bases. About 94 percent were either dead or injured to some degree. Only about 6 percent (5 of 84) had no obvious fire scars. Of the living fire-injured saguaros, nearly 80 percent (54 of 68) were completely girdled. This girdling typically occurred within 1 m above the soil surface (fig. 4). In addition, scorched, yellow-brown scars reached heights to 6 m on one or more sides of individual saguaros. These prominent scars (91 total scars on 54 saguaros), appear correlated with aspect: east (44 percent), north (36 percent), south (12 percent) and west (8 percent). Nearly half of these severely burned saguaros bloomed 1 year postfire (fig. 4) and provided a limited seed source.

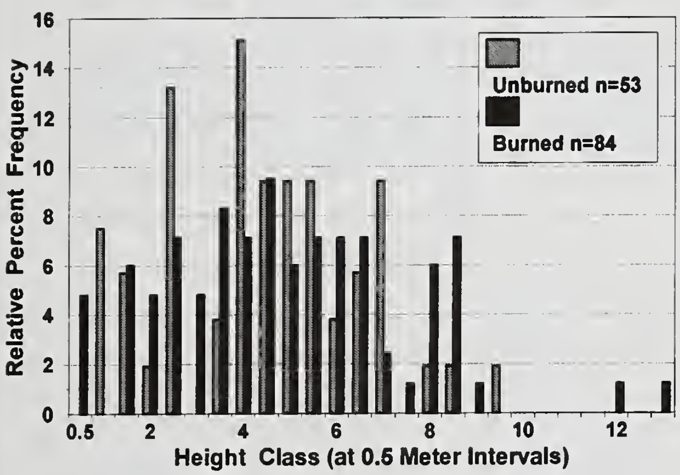

Figure 2. Relative percent frequency of saguaros, distributed by height class, from unburned and burned sample transects (total n=137) in and near the May 1993 Vista View Burn.

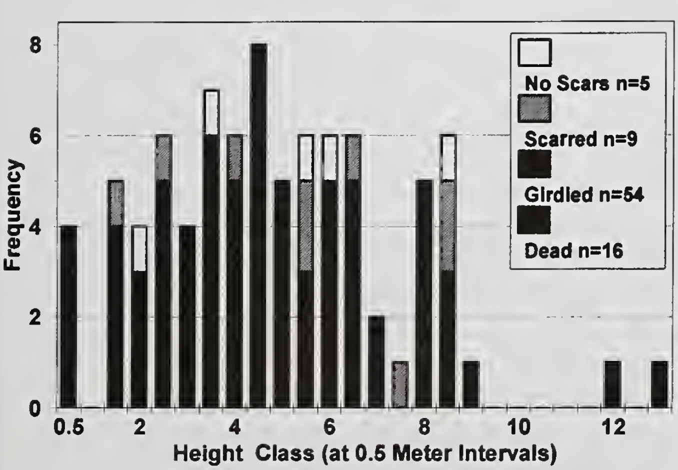

Figure 3. Frequency of wildfire injury to saguaros, distributed by height class, from sample transects (n = 84) in the May 1993 Vista View Burn.

Figure 4. Fire girdled (a) and scarred (b) saguaro blooming (c) 1 year after the May 1993 Vista View Burn.

In unburned areas 2 percent (1 of 53) of the saguaros were dead compared to 19 percent (16 of 84) in burned areas.

Injury, unrelated to fire, was observed on saguaros in unburned and burned sites. This injury was seen most often in the upper portions of the saguaros (e.g., bird cavities in the arms and upper trunk) and was clearly distinguishable from fire damage. Scars on saguaros in burned areas were obviously fire-caused and not the effects of epidermal browning.

Associated Vegetation

Six woody small-tree or large-shrub species, common to both unburned and burned areas, composed about 88 percent of the nearest neighbors to the 137 saguaros sampled. They included: foothill paloverde, *Cercidium microphyllum* (31 percent), white thorn, *Acacia constricta* Benth. (17 percent), wolfberry, *Lycium* spp. (17 percent), creosote bush, *Larrea tridentata* (DC.) Cov. (12 percent), catclaw, *Acacia gregii* A. Gray (6 percent), and crucifixion thorn, *Canotia holacantha* Torr. (5 percent). Other nearest neighbor shrubs, each with less than 3 percent frequency, included: mormon tea, *Ephedra trifurca* Torr., white ratany, *Krameria grayi* Rose & Painter, jojoba, *Simmondisa chinesis* (Link) C. Schneid, and *Yucca sp.* Of these, all except *L. tridentata* were resprouting in burned areas (Wilson et al. 1995a). *Carnegiea gigantia* occurred as a nearest neighbor twice in each of the unburned and burned areas.

Figure 5. Saguaro, *Carnegiea gigantea*, and its common nearest neighbor, foothill paloverde, *Cercidium microphyllum*, on the May 1993 Vista View Burn.

Cercidium microphyllum (fig.5) was the most frequent nearest neighbor to saguaros in both unburned (36 percent) and burned (27 percent) areas. Interestingly, it was also the most common nearest neighbor to the transect points (37 percent) in two burned areas.

About 9 percent of nearest neighbor plants were dead in both unburned and burned areas. Dead plants in unburned areas included: *C. microphyllum* and *Lycium* spp. Charred dead, non-sprouting plants in burned areas included: *C. microphyllum*, *Acacia* spp., *Ephedra* spp., and *Krameria* spp.

DISCUSSION

The succulent nature of saguaros makes them an unlikely fuel plant. Few, especially large individuals, appear to be consumed by fire. Yet, the fact that fire kills saguaros is documented here. Our findings corroborate those of Rogers (1985) and Thomas (1991) who both note the increased impact of fire in Sonoran Desert regions and high fire-caused cactus mortality, specifically for saguaro. Cave and Patten (1984) report a 100 percent fire-caused mortality for saguaro. Rogers (1985) reports 85 percent postfire saguaro mortality. Because these giant cacti hold large stores of both water and products from photosynthate, it may take several years before an individual dies from either frost or fire damage. Delayed mortality after fire may take over 6 years (Rogers 1985, Thomas 1991). We observed an initial postfire mortality of 19 percent, and expect most of the severely girdled saguaros to die prematurely. Based on this expected additional loss, our projection of over 80 percent eventual saguaro mortality from the Vista View Burn is substantial.

Regeneration of this saguaro population may depend on a diminished seed pool from the few surviving saguaros. Injured saguaros often continue to produce seeds (fig. 4) for several years (Rogers 1985). However, seedling success is linked to precipitation and temperature fluctuations, animal foraging, and

available suitable plant cover (Steenbergh and Lowe 1983). Postfire resprouts of associated vegetation (Wilson et al. 1995a) and reproduction by moribund saguaros may provide the potential for this saguaro-shrub community to recover after fire. However, intermittent livestock grazing may reduce seedling survival. Numbers of saguaros under 20 years of age are markedly greater in populations where livestock are excluded (Abouhaidar 1992).

Cyclic fire events may now influence the dynamics of major components in this saguaro-shrub ecosystem. Introduced grasses now supply a carpet of contiguous fuels that may contribute to larger conflagrations. Increased use of this National Forest increases the chance for more ignitions. Therefore, extra fire suppression efforts may be required to protect the saguaro resource, especially after seasons with high precipitation. Studies are needed to determine how fuels relate to fire-caused injury or survival. These studies should include analysis of:

1. Fuel loading,
2. Size, distance from, and aspect of nearest neighbors,
3. Aspect of greatest fire injury and
4. Weather conditions relating to fire behavior.

Postfire evaluations of saguaro injury and death should take into account any possible prefire damage from other factors. Evans et al. (1992) describe epidermal discoloration and spine loss on saguaros that could be mistaken for fire-caused injury.

More questions than answers remain. Can temporal burning reduce seed set of exotic herbaceous fuels and disrupt fuel contiguity without damaging saguaros? Prescribed burning as an alternative may increase biomass at the expense of desirable species such as the saguaro (Cave and Patten 1984). In high fire risk areas, can selective removal of flashy fuels save valuable individual cacti? Using grazing to reduce fuel buildup further reduces the chances that young saguaros may survive (Abouhaidar 1992). How can we reduce fire risks without losing the benefits of protection from frost and insolation provided to saguaro by its associated plants? Clearly, fire management alternatives are needed to reduce further degradation of this Sonoran Desert "keystone" species. Will the public support an unseen resource? Once lost, several human generations may pass before the giant saguaro regains its visible majestic status.

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Fire History in the Gallery Pine-Oak Forests and Adjacent Grasslands of the Chiricahua Mountains of Arizona[1]

Mark Kaib, Christopher H. Baisan, Henri D. Grissino-Mayer, and Thomas W. Swetnam[2]

Abstract.—Many authors have speculated about presettlement fire frequencies in semidesert grasslands and the relative importance of fire in the ecology of these systems. Yet, lack of direct evidence (e.g. fire scars) has hampered attempts to reconstruct the role of fire in these areas. Tangible evidence, however, is available from fire-scarred pines in canyon-gallery forests of the Madrean Province that recorded surface fires spreading from adjacent grasslands and savannas. Given the highly dissected topography that typically separates canyons, it is likely that many fires spread primarily into and between canyons from the lower savanna/grasslands, as opposed to originating at higher elevations. Inter-canyon synchrony of fire dates would provide supporting evidence for this hypothesis. Conversely, asynchrony would suggest lack of fire spread between canyons. If true, fire frequencies recorded in these gallery forests should approximate the minimum fire frequencies sustained in the adjacent grasslands.

Fire intervals at the canyon sites range between 3.0 and 4.0 years, while the synchronous intercanyon intervals range between 7.4 and 8.1 years. The range of fire frequency in the semidesert grasslands, given our interpretations, falls approximately between the individual canyon site fire frequency of four years, and the intercanyon fire frequency of eight years.

INTRODUCTION

The Madrean Province is a biome of disjunct mountain islands that connect the Southwest Borderlands of Arizona and New Mexico to the Mexican states of Sonora and Chihuahua (See Fig. 1). A biophysical and sociopolitical corridor linking the Southern Rockies and the Northern Sierra Madres, the Madrean Province is an "archipelago" of conifer-topped mountain islands surrounded by a "shore" of evergreen oak woodlands and a "sea" of semidesert grasslands (Gehlbach 1981; Wilson 1995). In this province, fire is widely regarded as a fundamental ecosystem process, yet, knowledge of past fire regimes in the pine-oak forests, oak woodlands, and semidesert grasslands is relatively limited. For example, many authors have used indirect historical evidence to infer presettlement fire frequencies in woodlands, savannas, and grasslands (e.g., Bahre 1995a, 1985; Baisan 1990; Baisan and Swetnam 1990; Hastings and Turner 1965; Humphrey 1963, 1958, 1984; Leopold 1924; McPherson 1995; Swetnam and others 1992, 1989). However, these findings encompass a broad variety of conclusions due largely to a lack of direct quantitative evidence of the role of fire in these communities.

Lightning and fire records from the Coronado National Forest demonstrate that, although lightning ignitions concentrate on mountain peaks, they are not uncommon in the foothills and grasslands (Baisan 1990; Baisan and Swetnam 1990; Barrows 1978). Regardless of the point of origin, given low fuel moisture conditions, rapid fire spread, and topographic position, once fires enter the grasslands, they typically spread quickly over large areas. Newspa-

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per and historical accounts from the late 1800s, for example, refer to "millions of acres" of grasslands and woodlands burning between May and July (Bahre 1985, 1991; Humphrey 1958).

The ecosystem management initiative (USDA 1992, 1993a, 1993b) has sparked interest in the ecological role of fire in the grasslands, savannas, and woodlands and its eventual use to emulate presettlement forest conditions and to restore or sustain biodiversity and productivity (Allen 1994; Allen et al. 1995; Kaufmann and others 1994). Fire history information obtained in canyon-gallery forests, adjacent to the semidesert grasslands, will provide tangible evidence (i.e. fire-scar chronologies) of grassland fire frequencies to support the reintroduction of fire in these borderland ecosystems. The overall objective of this project is to reconstruct the fire histories for several pine-oak forest sites adjacent to semidesert grasslands. Although additional sampling is planned, six collections of fire-scarred samples were obtained from sites in Rhyolite, Pine, Turkey Creek, and Rucker Canyons in the western Chiricahua Mountains, McClure Canyon in the northeast Huachuca Mountains, and Caj n del Oso in the northern Sierra de los Ajos of Sonora, Mexico (Fig. 1).

Data from Pine and Rhyolite Canyons (Swetnam and others 1989, 1992) are presented here and compared to a higher elevation site at Rustler Park

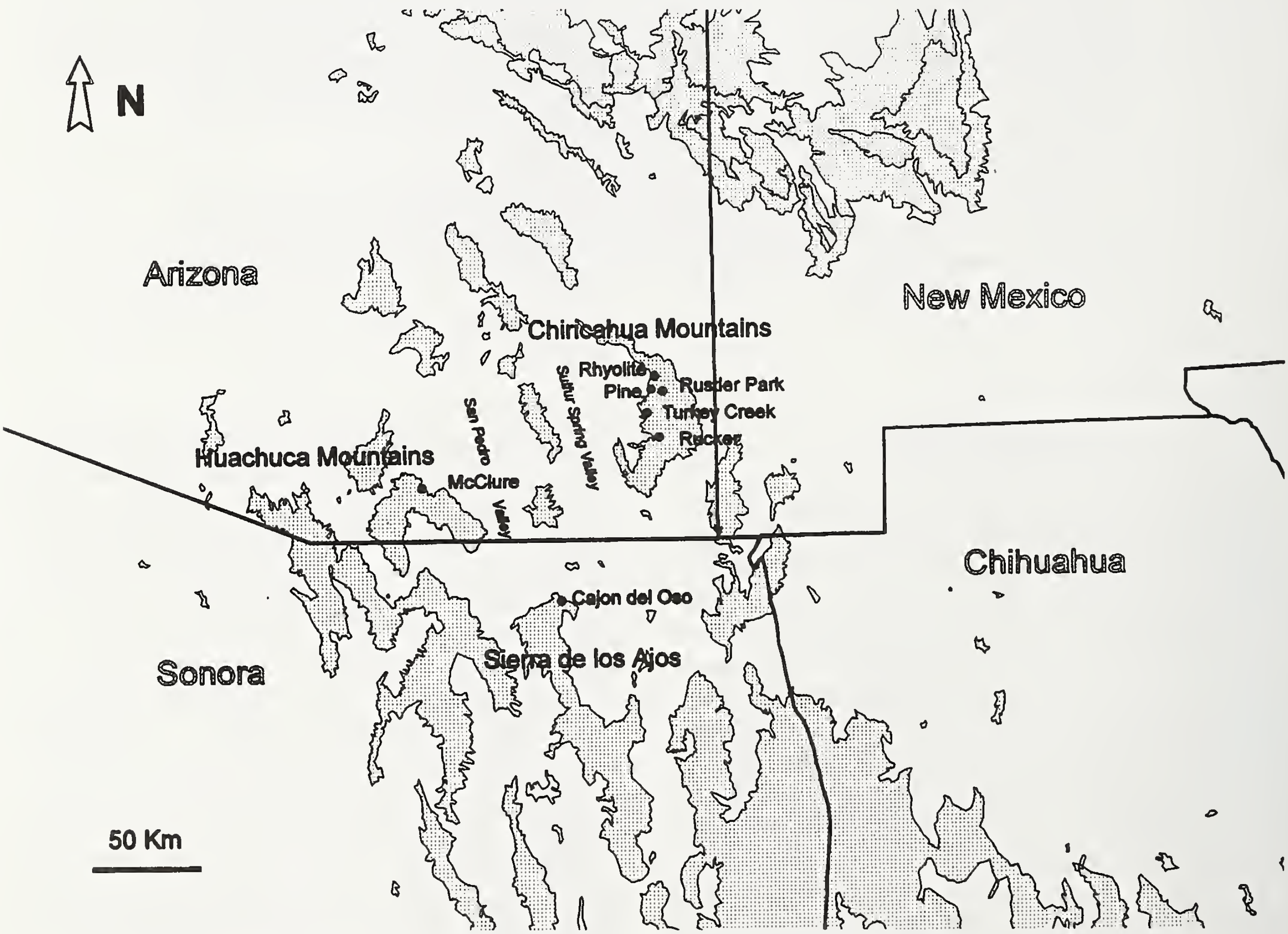

Figure 1. Madrean Province gallery-pine oak forest fire history reconstruction sites for the Southwest Borderlands project. Sites include Rhyolite, Pine, Turkey Creek, and Rucker Canyons within the Western Chiricahua Mountains, McClure Canyon at Fort Huachuca Military Reserve, and Cajón del Oso, in the Sierra Ajos, Sonora, Mexico. Map modified from Marshall (1957) by Bennet and Kunzmann (1992).

(Seklecki and others, this volume). We propose that fires recorded in these gallery forests provide strong evidence for the range of fire frequencies sustained by the adjacent grasslands. Given the highly dissected topography and the geomorphic and vegetative barriers separating these gallery forests, we hypothesize that fire spread primarily between canyons through the lower savanna/grasslands, as opposed to having spread from higher elevations. Intercanyon synchrony of fire dates would support this hypothesis and therefore historical fire frequencies in gallery forests provide a conservative (i.e., minimum) estimate of fire frequencies sustained in the lower semidesert grasslands.

HISTORICAL ECOLOGY

The rich cultural history of this region in context with fire reconstructions, in some cases, may account for anomalous patterns in past fire regimes. Fire histories in the Southwest, as a whole, show close association to regional climate reconstructions (Swetnam 1990; Swetnam and Baisan 1995) although individual sites occasionally exhibit unique fire patterns and characteristics attributable in some cases to anthropogenic effects (Baisan and Swetnam 1995). Early historic Amerind cultures that inhabited this region include the Sobaipuri (Pima), and later the Suma, Concho, Janos, Jacomes, Manos, and Apache (Bolton 1919; DiPeso 1929; Forbes 1966, Opler 1941, 1969). The Chiricahua Apache, the southern-most Athabascan group of hunters and gatherers, entered the province by at least the late 1600s (Aschmann 1956; Bolton 1967; Worcestor 1979) and raiding and warfare tradition resulted in conflict with Sobaipuri, then Spanish, Mexican, and later American regiments (Basso 1971; Castetter and Opler 1936; Naylor and Polzer 1986). Traditional use of fire by Apaches is suggested as a possible cause of anomalous fire patterns seen in certain locations of the borderlands area (Morino 1996; Seklecki and others, this volume).

Most historic ethnographic accounts of anthropogenic fire use in the Southwest Borderlands correspond with wartime periods (Bartlett 1954; Bourke 1887/88; Cole 1988; Sweeney 1991). These tentative wartime periods (late 1600s, 1748-1786, and 1831-1872) coincide with intervals of above average fire frequencies in the Chiricahua Mountains that may be associated with warfare tactics carried out by, and perhaps against the Chiricahua Apache (Kaib and others 1996). The temporal and spatial influence of anthropogenic-enhanced fire regimes may be inferred through historic ethnographic accounts, together with climate and fire histories, fire-scar seasonality, and through the spatial analysis of such patterns. The scale of anthropogenic fire influence will be the topic of future work in this region.

Although the Apaches led by Geronimo did not surrender until 1886, the relocation of the Chiricahua Apaches from their reservation in the Chiricahua Mountains to San Carlos in 1876 marks the beginning of widespread colonization in the Southwest Borderlands by Euroamericans. Intrepid pioneers migrated with mixed herds of livestock, and by the late 1870s, ranching, mining, fuelwood cutting, and logging all contributed to the local economy. By 1880 the numerous ranches in Sulphur Spring Valley contained an estimated eighteen thousand cattle (Bailey 1994; Wagoner 1952, 1961). The completion of the Southern Pacific Railroad in 1881 led to rapid immigration by ranchers with herds of thousands of cattle attracted to the lucrative grasslands and open ranges of Arizona (Wagoner 1961).

Early descriptions of the semidesert grasslands can provide useful information on the past form and function of fires in these systems. In his journals from the Southwest Borderland Apache campaigns in the 1880s, Captain John Bourke boasts: "as for the grasses one has only to say what kind he wants, and lo! It is at his feet—from the coarse sacaton which is deadly to animals except when it is very green and tender; the dainty mesquite, the bunch, and the white and black gramma, succulent and nutritious..... I must say, too, that the wild grasses of Arizona always seemed to me to have but slight root in the soil, and my observation is that the presence of herds of cattle soon tears them up and leaves the land bare" (Bourke 1891/1971, p. 140). Bourke witnessed the land degradation that followed the overstocking of the common-held rangelands in the 1880s (Bahre 1991; Hastings 1959).

Thirty years earlier, John Bartlett recorded fire effects on mesquite in his borderlands "explorations and incidents" (1850-1853, p. 75): "Where the prairies are frequently burned over, the tree is reduced to a shrubby state, a great number of small branches proceeding from one root, which goes on developing and attains a great size...... These roots , dug up and dried, are highly prized for fire-wood." Bartlett (1850-

1853, p. 75, 186, 344) notes on several occasions "the scarcity of firewood" in the semidesert grasslands and their dependence on mesquite roots for fuel. In the past mesquite was uncommon due to frequent fires, although present conditions are different, and in places of the San Simon, Sulphur Spring and San Pedro Valleys, large mesquite thickets are not uncommon (Bahre 1995b; Buffington and Herbel 1965; Hastings 1959). Although fires in settlement days were common, their effects were short lived and often beneficial as illustrated by this report in the Arizona Daily Star, Sept. 2, 1880: ...the grass over areas that were burned over this season is now knee high and looks as fresh as spring time in this locality [Patagonia]....(Bahre 1991).

The rapid rise in mining, logging, and primarily the livestock industry in the 1880s, all contributed to the landscape fragmentation and disruption of fuel continuity which severely limited fire spread (Bahre 1995a). The frequent surface fires which these grasslands and forests had sustained for several centuries, as illustrated by our fire history reconstructions (see fig. 2 and 3), were no longer recorded by pine trees following this period. Subsequent fire suppression policies (Swetnam and Baisan this volume) further altered the character and function of fire in these ecosystems (Baisan and Swetnam 1994; Leopold 1924; Marshall 1962). A century of fire exclusion has resulted in a dramatic transformation in ecosystem structure (Cooper 1960; Weaver 1951; Marshal 1957, 1963), and a concurrent shift from frequent surface toward infrequent uncontrollable stand replacing fires. The benign fires which commonly swept across this region, following a century of fuel accumulation, are now different in nature (i.e. 1994 Rattlesnake Burn, Chiricahua Mountains) and likely outside the natural range of fire regime variation.

SITE DESCRIPTION

Rhyolite and Pine Canyons are two of several intermittent stream systems that drain the western slopes of the Chiricahua Mountains. Ephemeral streamflow, perennial baseflow, topographic sheltering from wind and sun, and cold air drainage all contribute to the relatively mesic conditions that harbor these unique gallery pine-oak forests. Major plant associations found dispersed throughout these rhyolitic canyons include pine-oak forest, southwest riparian deciduous forest, madrean oak woodland, interior chaparral, plains grassland, and semidesert grassland (Brown 1982; Brown and Lowe 1980; Marshall 1957; McClaran 1995; Wallmo 1955). These canyon-gallery forests occupy a topographic niche (1800 - 2500 meters) between the lower oak woodlands and semidesert grasslands and the upper ponderosa pine forests. Major tree species found at the canyon sites include; Apache pine (*Pinus engelmannii*), Chihuahua pine (*P. leiophylla var. chihuahuana*), Arizona pine (*P. Ponderosa var. arizonica*), border pinyon (*P. discolor*), Douglas-fir (*Pseudotsuga menziesii*), Arizona madrone (*Arbutus arizonica*), Arizona cypress (*Cupressus arizonica*), alligator-bark juniper (*Juniperus deppeana*), Arizona sycamore (*Plantanus wrightii*), Arizona walnut (*Juglans major*), silver-leaf oak (*Quercus hypoleucoides*), Arizona white oak (*Q. arizonica*), Emory oak (*Q. emoryi*), and netleaf oak (*Q. rugosa*; Barton 1994; Bennett and others, in Prep; Reeves 1976).

METHODS

Partial and full cross sections from fire-scarred Apache and Arizona pine stumps and logs were obtained using a chain saw along a 600 meter elevation gradient in Pine Canyon. All samples were finely polished with a belt sander, then used to reconstruct a multi-century fire history (Arno and Sneck 1977; Dieterich 1980, 1983; Dieterich and Swetnam 1984; Weaver 1951) using dendrochronological (tree-ring dating) methods (Stokes and Smiley 1968; Swetnam and others 1985). Using statistical and graphical analyses (Grissino-Mayer 1994) we compared the Pine Canyon fire history reconstruction with one developed for Rhyolite Canyon in Chiricahua National Monument (Swetnam and others 1989; 1992), and one developed for a higher elevation ponderosa pine/mixed conifer site at Rustler Park (Seklecki and others, this volume). The period 1700 through 1876 was chosen for analyses, because adequate 18th century sample depth (fire-scarred tree numbers) existed for all sites and 1876 was the last fire event recorded at Pine Canyon (see fig 2 and 3). The 1700-1876 period was analyzed for all fires recorded and all fire dates recorded by two or more trees.

The mean and median fire intervals are considered best measures of central tendency in symmetric

Pine Canyon, Chiricahua Mountains, Master Fire Chronology

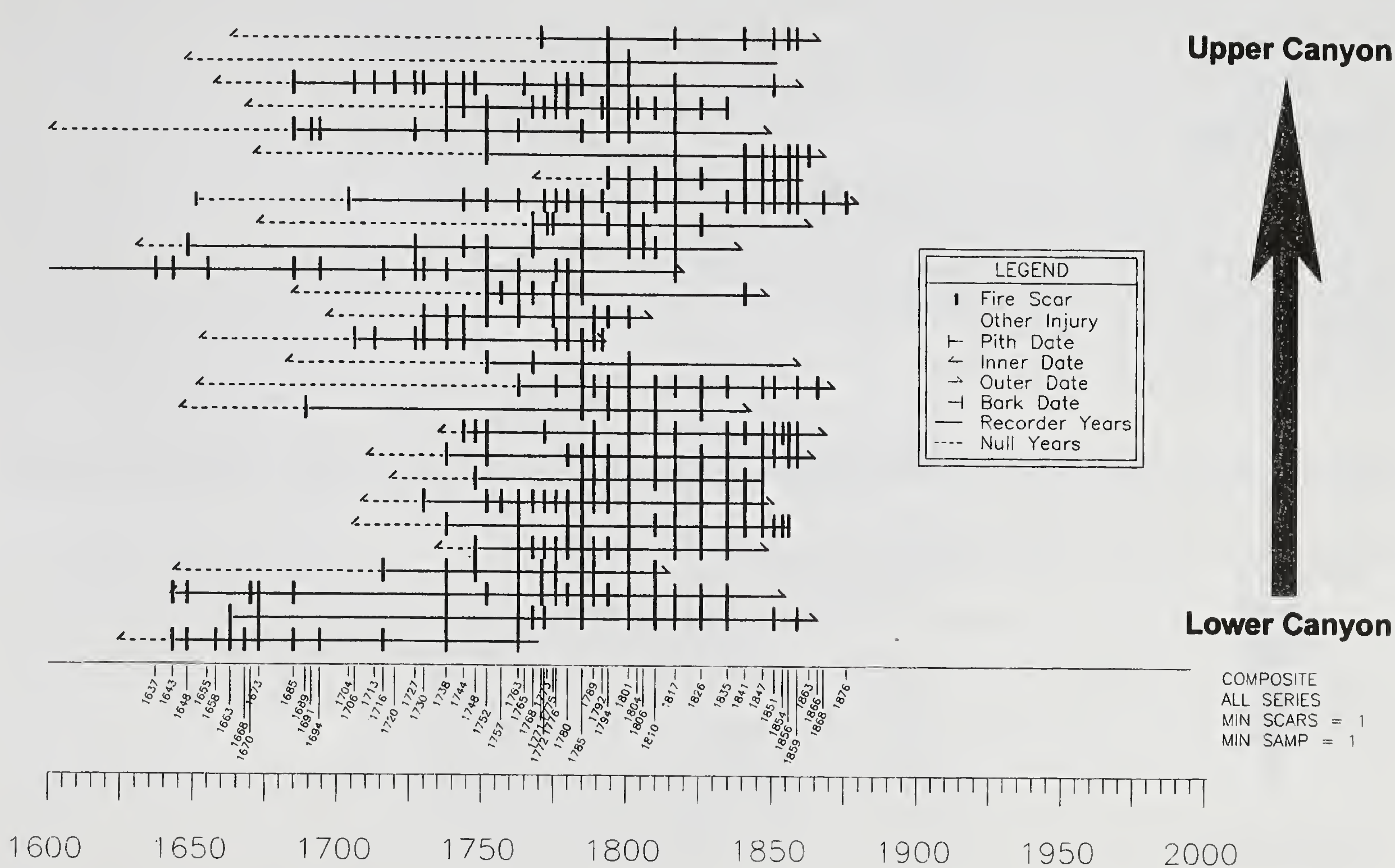

Figure 2. Master fire chronology for Pine Canyon. Horizontal lines represent information from individual trees while vertical bars represent dated fire events (scars).

Rhyolite Canyon, Chiricahua National Monument, Master Fire Chronology

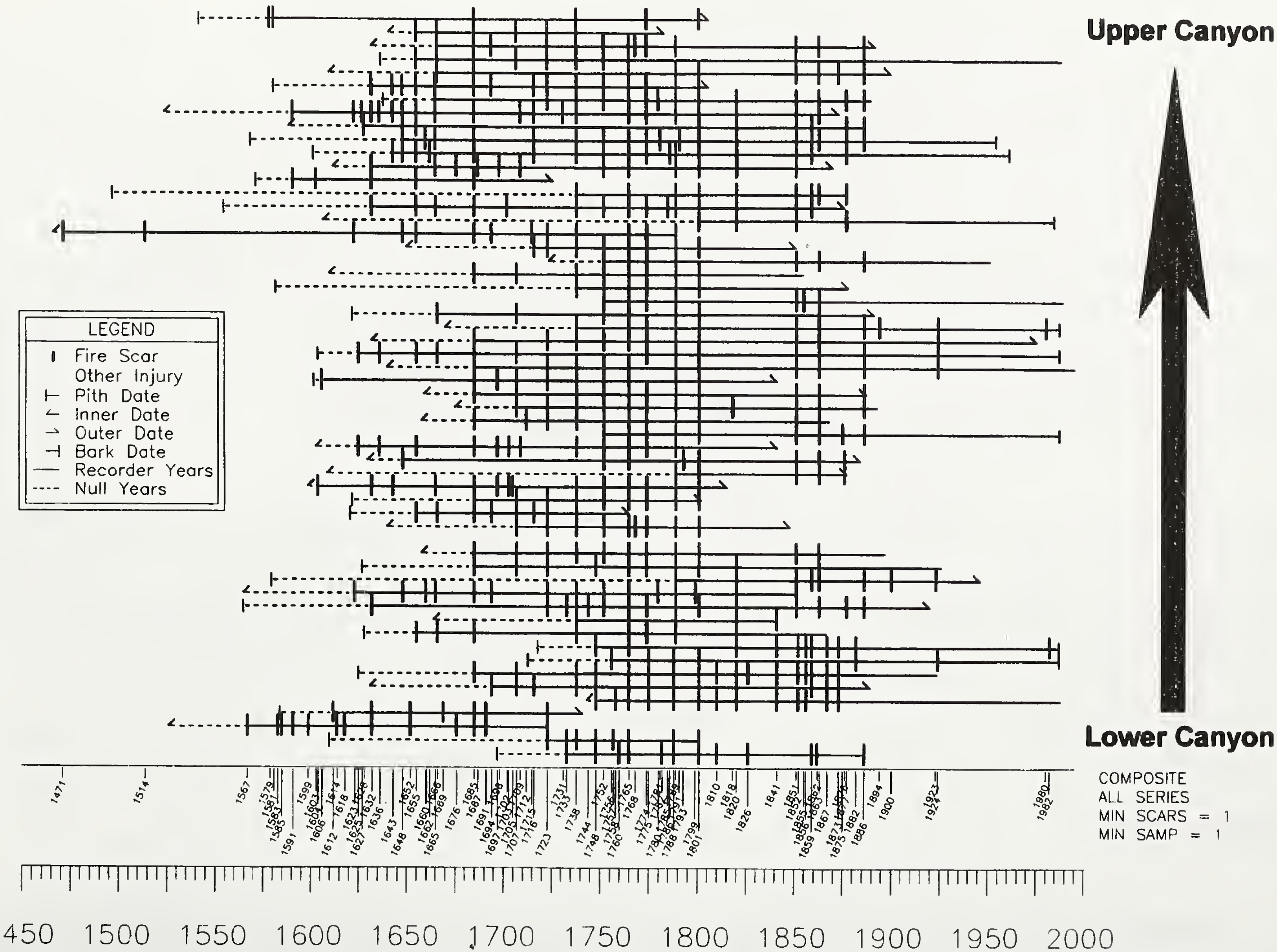

Figure 3. Master fire chronology for Rhyolite Canyon. Horizontal lines represent information from individual trees while vertical bars represent dated fire events (scars).

distributions while the Weibull median probability interval (WMPI; Grissino-Mayer 1995; Weibull 1951) is considered a more robust measure for non-normal distributions. As fire interval data are typically positively skewed, the WMPI is considered a more robust measure of central tendency in fire interval distributions (Grissino-Mayer 1995). Our estimates of fire frequency are based on the least conservative measures of central tendency (i.e. all fires recorded and smallest WMPI) due to the limited samples available at these sites, and the possibility, that all fires were not recorded by sample collections.

All sites were analyzed for spatial patterns of past fire and non-fire event association using a 2 x 2 chi-squared test and Yule similarity index (Grissino-Mayer 1995; Swetnam 1993). The 2 x 2 chi-squared test (Ludwig and Reynolds 1988) is used to test association between two sets of binary data (+,- ; presence, absence) based on four cells (++, +-, -+,—). The 2 x 2 test is analogous to a two coin toss under the null hypothesis that the pattern of fire and non-fire years is statistically independent between sites (Grissino-Mayer 1995). The Yule index (Hubalek 1982) is a measure of association between two sets of binary data and ranges from 0.0 at minimum association to 1.0 at maximum association, with values below 0.5 having little statistical association and those above having proportionately greater association.

RESULTS

A fire history reconstruction was compiled for each canyon site and for the common intercanyon fire dates (Fig 2, 3, and 4). Comparison of fire history reconstructions, fire frequency, and descriptive statistics (Fig 2, 3, and 4; Table 1) reveal that the higher elevation Rustler Park site sustained more frequent fire than the canyon sites. This is not surprising as the Rustler Park site is situated at the mountain crest and accessible to fire spreading from numerous corridors on both sides of the mountain. Between 1700 and 1876, 30 out of 73 fire years (41 percent) were synchronous between Pine Canyon and Rustler Park (separated by 4.8 km). However, only 28 out of 83 fire years (34 percent) were synchronous between Rhyolite Canyon and Rustler Park (12.8 km), while only 21 out of 71 fire years (30 percent) were synchronous between Rhyolite and Pine Canyons (6.4 km). Al-

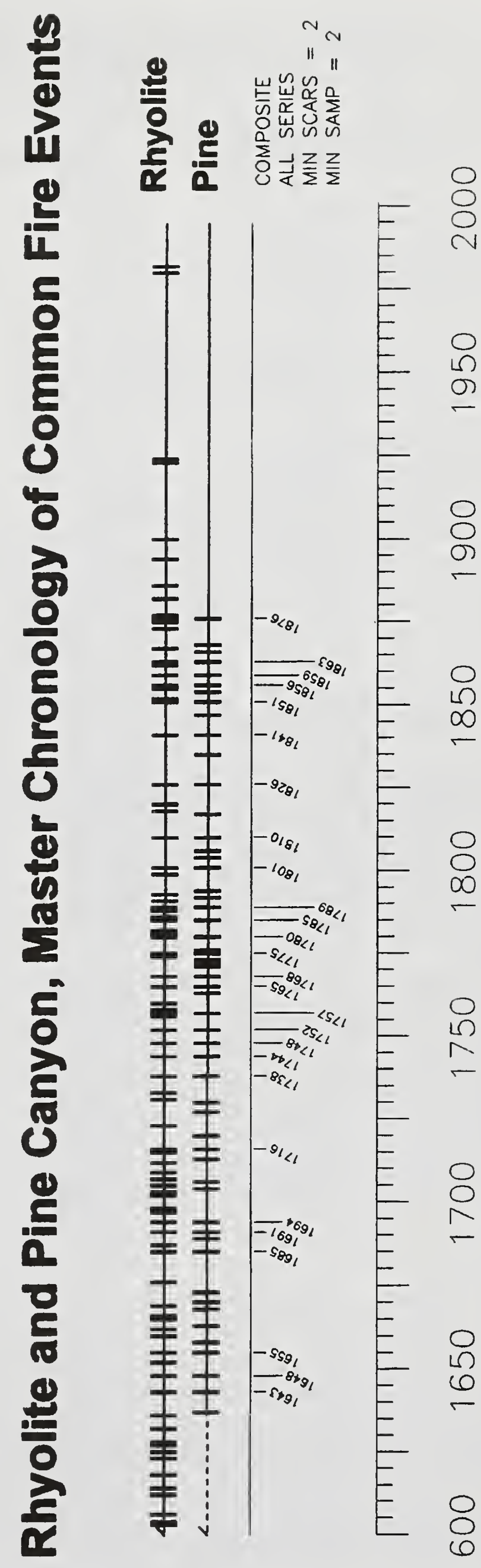

Figure 4. Composite fire chronology of fire-scar dates held in common by Pine and Rhyolite Canyons and interpreted to be grassland-spread fires. Horizontal lines represent fire composites from individual canyons while vertical bars represent the aggregate of fire dates from each site.

Table 1. Descriptive fire statistics for Rhyolite Canyon, Pine Canyon, paired canyons, and Rustler Park sites between 1700 - 1876 for all fire dates, and fire dates recorded by two or more trees. The paired analyses includes the common Pine and Rhyolite fire events.

All Fire Dates Recorded	Pine	Rhyolite	Paired	Rustler
Total Fire Dates	42	50	70	61
Mean Fire Interval (yrs.)	4.2	3.6	2.5	2.9
Median Fire Interval (yrs.)	4.0	3.0	2.0	2.5
Weibull Median Probability Interval (yrs.)	4.0	3.0	2.2	2.7
Interval Range (yrs.)	1-9	1-15	1-9	1-16
Fire Dates Recorded by > 2 Trees	**Pine**	**Rhyolite**	**Paired**	**Rustler**
Total Fire Dates	33	29	20	50
Mean Fire Interval (yrs.)	4.8	6.2	8.0	3.4
Median Fire Interval (yrs.)	4.0	5.5	5.5	3.0
Weibull Median Probability Interval (yrs.)	4.6	5.6	7.4	3.2
Interval Range (yrs.)	1-11	1-15	3-22	1-16

though only 16 out of 90 fire years (18 percent) were synchronous between all three sites, of the 21 fires years in common between Pine and Rhyolite Can 16 were also common to Rustler Park (81 percent). This suggests that a high portion of the common intercanyon fires (fires which likely spread between the canyons through the grasslands) also spread to Rustler Park, and we would likewise infer these to be larger-scale fire events. These large fire years are well represented in the paired-canyon fire chronology (Fig. 4).

Values for the WMPI range between 3.0 and 4.0 years for all fire dates recorded, and between 4.6 and 5.6 years for all fire dates recorded by two or more trees (Table 1). Because of numerous potential barriers to fire spread at the canyon heads, and in surrounding uplands, we interpret synchronous intercanyon fire-scar dates as fires that most likely spread between the canyons through adjacent grasslands. Therefore, the common intercanyon fire reconstruction provides a reasonable estimate of the temporal patterns of semidesert grassland fires over several centuries (Fig. 4). The common intercanyon values for the WMPI range between 7.4 and 8.1 years for all fire dates recorded by two or more trees (Table 1). Therefore, a conservative estimate of intercanyon fire frequency (i.e., grassland spread) would be 8 years. The range of fire frequency in the semidesert grasslands, given our interpretations, should fall approximately between the individual canyon site fire frequency of 4 years, and the intercanyon fire frequency of 8 years (Table 1). Hence this data suggest a fire frequency in these semidesert grasslands on the order of 4 to 9 years. Because limited fire-scarred samples are preserved in these gallery forests, portions of the multi-century record are incomplete resulting in a conservative estimate of fire frequency. It is possible that the semidesert grassland fire frequency is closer to the shorter estimate of 4 years.

All chi-squared tests were significant ($p < .005$) between the sites (Table 2). Therefore, we conclude that the patterns of fire and non-fire years were not statistically independent between sites. This is not surprising considering the relative proximity between sites, the potential for intercanyon fire spread, and the influence of regional climate. The values of the Yule index further illustrate the degree of association between sites (Table 2).

Table 2. Chi-squared (2 x 2) paired testing of synchronous fire and non-fire events between three sites in the Chiricahua Mountain. All tests were significant ($p < .005$).

Sites for period 1700 - 1876		Rhyolite	Rustler Park
Pine	2 x 2 chi-squared	12.85	33.31
	Yule Index	0.64	0.72
Rustler Park	2 x 2 chi-squared	14.31	—
	Yule Index	.64	—

CONCLUSIONS

The range of fire frequency in the semidesert grasslands falls approximately between the individual canyon site fire frequency of four years, and the intercanyon fire frequency of eight years. Because these multi-century fire reconstructions represent a conservative estimate of fire frequency, we believe that the semidesert grassland fire frequency is probably closer to the shorter estimate of 4 years.

Reconstructed fire histories in the gallery-pine oak forests of Rhyolite and Pine Canyons provide multi-century illustrations of the fire regime patterns in these canyon systems. Additionally they provide evidence of fire frequencies sustained in the adjacent semidesert grasslands. Through the 1600s, 1700s, and 1800s, fires in these canyon forests, and probably also the adjacent grasslands, burned on the order of once every four to eight years. Although we attribute intercanyon fire synchrony to grassland fire spread we would expect some degree of fire synchroneity due to regional climate influence (Swetnam and Baisan 1995). The influence and scale of regional climate-, anthropogenic-, and grassland-fire effects can be estimated through rigorous analysis of previously developed and forthcoming fire chronologies in the Southwest Borderlands, and will be the ambition of future work. Additional chronologies, presently being developed in the Southwest Borderlands, will substantially expand the spatial inference of fire size and spread, and increase our understanding of fires role in these areas. This knowledge will provide land managers in the Borderlands region with three centuries of fire-regime patterns across several vegetation zones for use in education, guidance, and support in fire planning.

This research illustrates the essential function and character of fire, and the changes associated with its removal from these systems. Prescribed fires provide a viable ecological and economical tool that can be used to bring these systems back to presettlement conditions while concomitantly sustaining diversity and productivity. Although, before natural fire conditions can be emulated, planned small scale stand-replacement burns, fall and early spring burns, and various fuel management options should be considered to initially break the forest homogeneity, reduce catastrophic fire risk, and to ultimately restore these systems to a more productive, diverse, and sustainable state.

Future borderland fire history research will increase our knowledge and understanding of fire size and frequencies sustained by the pine-oak forests and lower semidesert grasslands in this region. Crossborder comparisons of fire histories will allow for evaluation of the effects of land use patterns, such as livestock grazing, logging, fuelwood harvesting, and fire suppression, on landscape fire patterns (Savage and Swetnam 1990; Touchan and others 1995). These findings will be directly useful for planning fire management programs aimed to restore fire to these ecosystems at intervals and seasons appropriate for maintaining biological productivity and diversity (Swetnam and Baisan, this volume). Additionally, they will provide managers with information for characterizing the presettlement "range of variability" of fire frequency for these unique systems across local to regional scales (Allen 1994; King 1995; Morgan and others 1994; Swanson and others 1994).

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The Influence of Fire and Land-use History on Stand Dynamics in the Huachuca Mountains of Southeastern Arizona

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Abstract.—Dendrochronological methods were used to reconstruct fire regimes and stand age structures in the Huachuca Mountains of Southeastern Arizona. Pre-settlement (i.e., before ca. 1870) fire intervals ranged from 4 to 8 years, with many fires spreading over the entire sample area. Stand age distributions show an increase in more shade-tolerant tree species. Although ponderosa pine still dominates stands, recent recruitment is predominantly southwestern white pine and Douglas-fir. Establishment of Ft. Huachuca in 1877 was a precursor to extensive use of timber, mineral, range and water resources in the Huachuca Mountains. The fire regime was clearly altered at this time, with only one subsequent widespread surface fire recorded in 1899. Settlement era land-use practices may be responsible for changes in stand structure and composition.

INTRODUCTION

It is widely thought that current conditions in most ponderosa pine *(Pinus ponderosa* Dougl. ex Laws.) forests are radically different than those that existed during pre-settlement times, i.e., prior to extensive Euro-American settlement in the late 1800s (Cooper 1961; Covington and Moore 1994). In particular, changes have been noted in forest structure (i.e., tree density, age distribution, species composition). Many forest stands in southern Arizona now contain an understory of dense, young ponderosa pines, as well as greater numbers of Douglas-fir *(Pseudotsuga menziesii* (Mirb.) Franco) and southwestern white pine*(Pinus strobiformis* Engelm.). These changes in structure and composition are a result of several historical factors, including decreased frequency of widespread surface fires, effects of human land-use, and variation in overall patterns of climate. Several studies in the Southwestern U.S. quantitatively described existing size and age structure of selected ponderosa pine stands (Pearson 1950; Cooper 1960; Schubert 1974; Savage 1991; White 1985; Fule and others 1995). Many of these studies point out that changes in forest dynamics are a result of both endogenous and exogenous factors, but few of them explicitly investigated these factors. Dendrochronology offers methods for simultaneously reconstructing tree age distributions, fire history, and climatic variations. By reconstructing stand history and making comparisons at different points in time we can quantify the extent of stand alterations and gain insight into the processes that created these changes.

This study was designed to characterize forest age structure and composition at the stand level and to use these data to improve our understanding of forest dynamics in the context of the historical fire regime and human land-use. Our sampling and analysis had two specific objectives. The first was to reconstruct the historical fire regime, including the occurrence, spatial distribution, and seasonality of past fires, over an elevational gradient using fire-scar analyses. The second was to determine the tree age structure and species composition in the area from which the fire-scarred samples were collected. Forest structure and composition were compared with the fire history, and with the land-use history derived from historical and current records of the site.

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SITE CHARACTERISTICS

The Huachuca Mountains are within the Madrean Archipelago or "sky islands" of southeastern Arizona. They are floristically diverse (Bowers and McLaughlin 1994; Wallmo 1955), as a result of the geographic conjunction of a variety of floristic types (e.g. Rocky Mountain, Cordilleran, Sonoran, and Chihuahuan), complex topography, and variation of precipitation and temperature with elevation. There is an upward transition from Sonoran Desert scrub at around 1,200 meters to mixed-conifer forest at over 2,800 meters. The study site is located at the juncture of the Garden Canyon and Ramsey Canyon watersheds (figure 1). Elevation of the area sampled for fire history ranges from 1,829 to 2,590 meters. Stand structure was sampled at the boundary between Fort Huachuca (FHA) and the Miller Peak Wilderness area (MPW) at 2530 meters elevation. Portions of the site are within the perimeter of the 1983 Pat Scott Peak fire and the 1980 Sawmill Canyon fire.

METHODS

Sixty-five full and partial cross-sections from fire-scarred logs, snags, and stumps, including partial cross-sections from 10 living trees were collected (Arno and Sneck 1977) along an elevational gradient from 1,830 to 2,500 meters. Approximately half of the samples were from the Garden Canyon watershed (Fort Huachuca) and the other half were from within and adjacent to the burned area centered around Pat Scott Peak in the Miller Peak Wilderness Area. This site was chosen due to ease of access, ability to compare sites across a management boundary, and because the site is fairly representative of the ponderosa pine forest in this mountain range.

Cross-sections were surfaced with progressively finer sandpaper using a belt sander to clearly reveal the wood structure, then crossdated. Crossdating compares patterns of narrow and wide rings between samples allowing for exact calendar years to be assigned to annual growth rings (Stokes and Smiley 1968). Once crossdated, the years in which fire scars were present were recorded for each individual tree (Dieterich and Swetnam 1984). Other characteristics recorded were intra-ring fire-scar position and occurrence of injuries by unknown factors (Baisan and Swetnam 1994). Thirty living trees were cored to develop a local tree-ring width chronology to facilitate crossdating samples with fire scars.

Statistical analysis was performed on the fire interval data using FHX2 software (Grissino-Mayer 1995) that analyzes fire history data both temporally and spatially. Three measures of central tendency were computed: mean fire interval (MFI), median fire interval and the Weibull median probability interval (WMPI - Grissino-Mayer 1995). Other parameters calculated, were the standard deviation, coefficient of variation, skewness, kurtosis, and minimum and

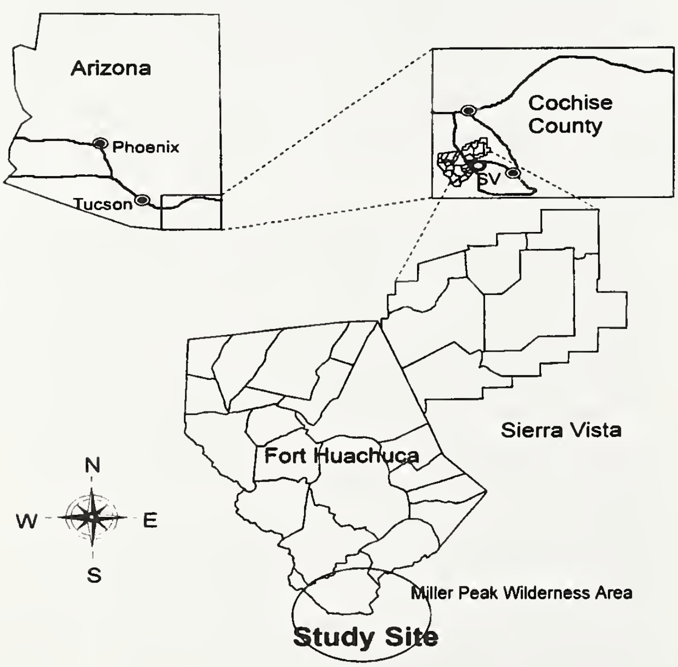

Figure 1—Location of study site at boundary between Fort Huachuca and the Miller Peak Wilderness area.

Table 1. Fire statistics using period of best sample replication 1689 to 1880. *ALL* is based on all fire events; *10%* (medium fires) and *25%* (large fires) with a sample depth of 3 or more trees scarred per fire event.

Statistic	All	10 pct.	25 pct.
Mean Fire Interval	4.5	4.9	8.4
Median Fire Interval	4.0	4.0	5.5
WMPI (Weibull Distribution)	4.4	4.7	7.6
Standard deviation	2.3	2.4	6.4
Coefficient. of variance	0.51	0.49	0.76
Skewness	1.1	0.94	1.7
Kurtosis	0.5	0.06	1.72
Minimum interval	1	1	3
Maximum interval	11	11	26

maximum intervals between fires (table 1). Season of fire occurrence was estimated by the relative positions of fire scars within the annual ring. The seasonal estimates were based on knowledge of cambial growth in southwestern Arizona conifers (Baisan and Swetnam 1994; Dieterich and Swetnam 1984; Grissino-Mayer *et al.* 1994; Fritts 1976).

Six 100 meter transect plots were established to sample the age and size distribution of the forest community within the area of the fire-scarred samples. Trees >12 centimeters in diameter were cored and tagged with a metal tag every two meters along the length of each transect. This sampling strategy was chosen to yield a sufficient number of samples representative of tree composition, ages, and sizes for this area (Avery and Burkhart 1994). Cores were mounted and finely surfaced to define ring boundaries, then crossdated against the established chronology. Height of core from ground surface, diameter at core height, species, and distance from transect were recorded for each tree.

RESULTS

Fire History

The tree-ring chronology, developed from the increment cores and cross-sections, ranged from A.D. 1499 through A.D. 1995 (497 years). Of the 65 cross-sections obtained, 57 were used to reconstruct the fire history and to analyze spatial distribution of major fires in the Garden Canyon Watershed. The remaining samples were not datable for various reasons such as severely reduced ring growth, decay, or anomalous ring patterns. Figure 2 is a master fire

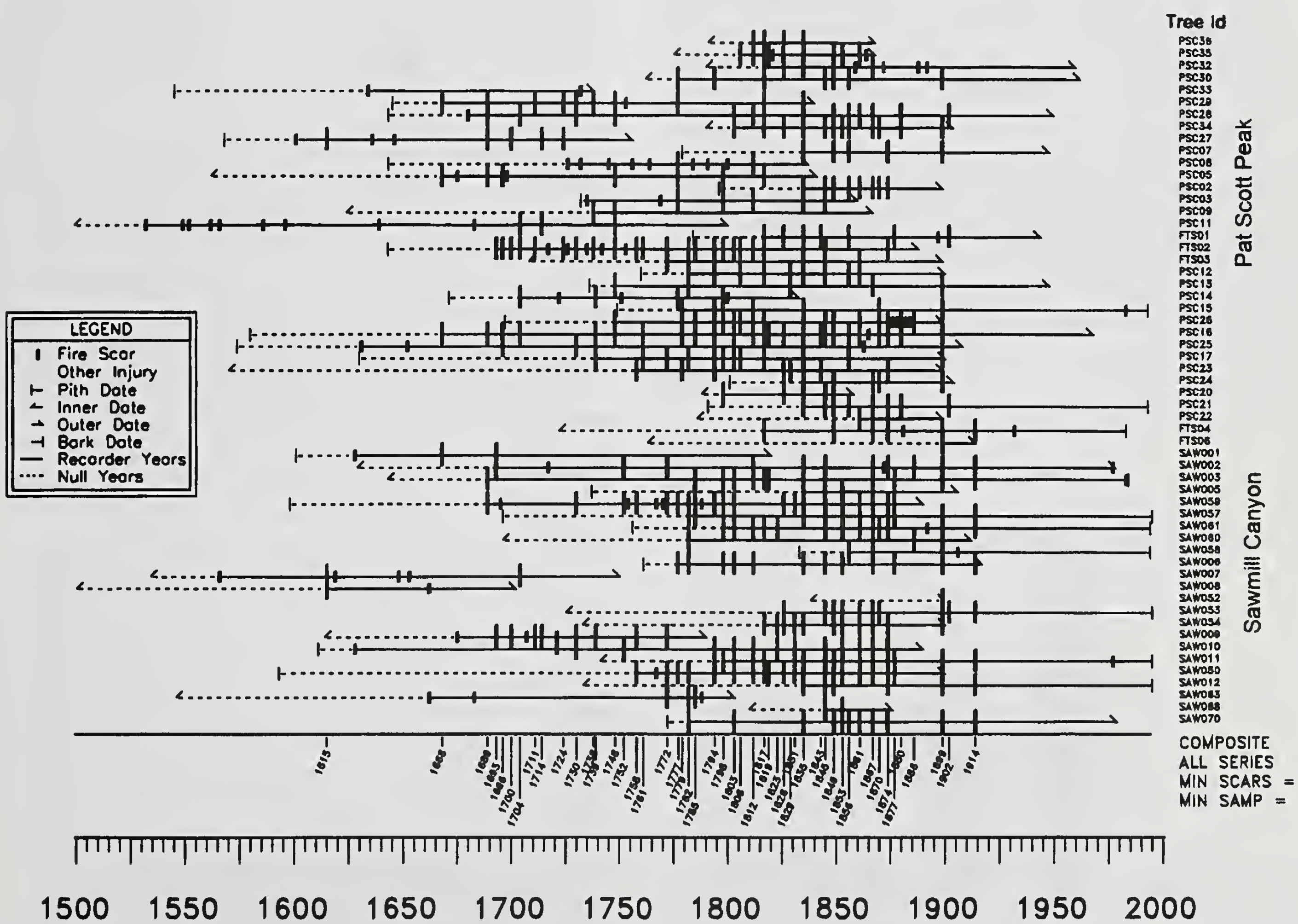

Figure 2. Master fire chronology of all trees, all fire events in the Garden Canyon Watershed. Each line represents a tree; each verticle line represents a fire scar.

chronology showing all trees with all fire years. As observed in many other Southwestern fire history studies, there is high synchrony among fire dates (Swetnam and Baisan, in press). Of 112 fire events, 38 percent were synchronous throughout the entire sample area (*i.e.*, fires were recorded by trees in both the upper and lower watershed), 44 percent were present in the upper watershed only and 18 percent in the lower watershed only.

Figure 3 shows the estimated patterns of seasonal fire occurrence. At least 90 percent of fires probably occurred before the summer monsoons began, typically before late June or early July.

The fire intervals during the analyzed time period were positively skewed. Because the mean is a less robust estimate of central tendency when the data are skewed, the data were modeled with the Weibull distribution. The Weibull is a flexible model designed to estimate means of non-symmetric data. Both the MFI and the WMPI show a range of 4 to 8 years. Table 1 describes summary statistics for the period of best sample replication 1689 - 1880. The "All" category includes all trees and all fire events. The 10 and 25 percent categories include only those fire events that were recorded by a minimum of 3 trees, thus excluding many of the fires that were probably less important in the study area or spatially patchy. Fire years in which 10 to 25 percent of the sampled trees were scarred are interpreted as intermediate-sized fires, and fire years in which greater than 25 percent were scarred are interpreted as relatively large fires.

Stand Structure

Tree recruitment patterns can be useful in determining effects of disturbances. Figure 4 shows a cumulative age distribution for each site by species. If recruitment and mortality rates were constant through time, this graph should, theoretically, be a straight line. However, the exponential shape of these curves empirically describes the observed tree ages, with few trees represented in older age classes and many trees in younger age classes. This pattern suggests a forest recovering from disturbance (Parker and Peet 1984), in this case the fire of 1899. Fort Huachuca transects (FHA) show more or less continuous recruitment of ponderosa pine (PIPO) since the turn of the century, with southwestern white pine (PIST), Douglas-fir (PSME) and Gambel oak *(Quercus gambelii* Nutt.)(QUGA) as minor components, with a pulse of recruitment around 1920. Miller Peak Wilderness Area (MPW) transects show a steep recruitment event of PIPO after 1890 and of PIST around 1930.

The largest pulse of recruitment for ponderosa pine in each site occurred between 1900 and 1910, possibly in response to the 1899 fire. From 1910 to 1950 however, southwestern white pine, Gambel oak, and Douglas-fir became well established, with recruitment matching that of ponderosa pine during the same period of time, probably in response to lack of fire, and to increased shading which favor these relatively shade-tolerant species.

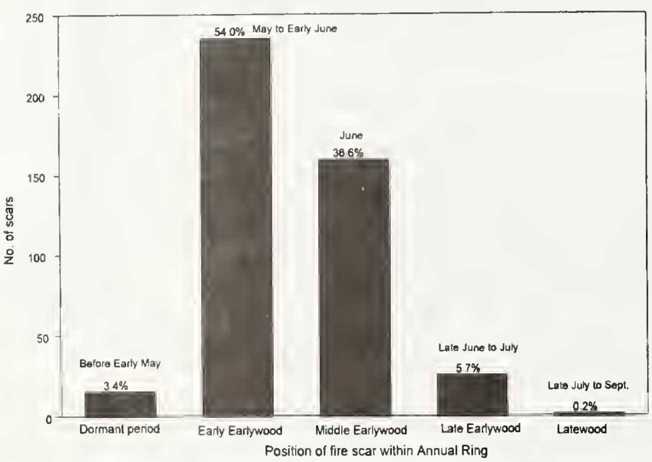

Figure 3. Estimated season of fire by identifying position of fire scar.

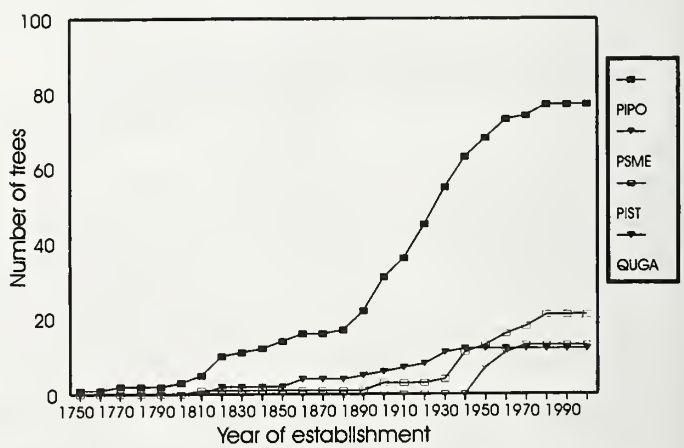

Figure 4. Cumulative 10-year age groups by species for each site (number of transects/site = 3).

DISCUSSION

The historical occurrence of widespread fires prior to 1880 suggests a continuity of fuels and few barriers to fire spread. The presence of multiple fire scars on many trees throughout the sampled area, and the presence of oaks many centuries old, are also an indication of widespread, low intensity fires. The 4 - 8 year mean fire interval is within the range of frequencies documented for similar studies in other Southwestern pine forests (Baisan and Swetnam 1990; Grissino-Mayer and others 1993; Swetnam and others 1992; Swetnam and Baisan, in press). The effects of past land use practices and fire suppression are reflected in these data. A pattern of very frequent fires of various sizes, both small and widespread, was interrupted in the late 1870s or early 1880s. Two widespread fires occurred after 1880, in 1899 and 1914. Following establishment of Fort Huachuca in 1877 there was extensive exploitation of timber, mineral, range and water resources in the Huachuca Mountains. The population of Euro-American settlers increased after this time, with the establishment of ranches on the plains surrounding the mountains, and development of sawmills and mines within the mountains (Bahre 1991; Hadley and Sheridan 1995).

Fuel consumption by the frequent surface fires most likely inhibited the occurrence of crown fires. In contrast to the low intensity pre-settlement fires, several large crown fires have occurred within the last 100 years in the Huachucas. The 1899 fire was the first in a series of severe, stand-replacing fires in the Huachucas that were probably associated with changes in forest structure. In the late 1800s logging was primarily carried out on "the Reef", a less precipitous area below Carr Peak. Logging was also conducted below Ramsey Peak on the military reservation. The 1899 fire terminated all large-scale logging operations (Wallmo 1955; Wilson 1995). Other catastrophic fires that occurred in the upper elevations of the Huachucas - Carr Peak in 1977 and Pat Scott Peak in 1983 - may have been due at least in part, to accumulation of fuels, both horizontally and vertically, leading to a cycle of infrequent, high-intensity fires.

Frequent fires before the 1880s served to keep fuel accumulation to a minimum and to keep stands relatively open by inhibiting establishment of less fire resistant species. Both the presence of Gambel oak and ponderosa pine individuals >200 years old on FHA suggest lower fire intensity and less human impact in this part of the watershed. While ponderosa pine is still the dominant tree species, lack of fire has probably encouraged growth of less fire resistant species such as Douglas-fir, Gambel oak, and southwestern white pine, and inhibited growth of ponderosa pine.

CONCLUSION

Fire is an important natural process that has influenced forest structural patterns and species composition in the Huachuca Mountains. During the last 100 years anthropogenic factors have also played an important role in structuring the forest community, promoting changes in the fire regime similar to those changes documented in other southwestern mountain ranges. The exclusion of fire has resulted in an encroachment of more shade-tolerant, less fire-resistant trees and a general decrease in the regeneration of ponderosa pine. Although areas within the study site that are still relatively open, (*i.e.*, wide grassy spaces between dominant ponderosa pine), other areas are becoming laden with fine fuels and needle cast, woody debris and multiple layers of fire-intolerant trees. Under the right climatic conditions or with careless human use of fire, these areas may be prone to another stand-replacing fire in the near future.

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Image Processing Techniques for Automated Terrain Stratification[1]

Michael J. Medler, Mark W. Patterson, and Stephen R. Yool[2]

Abstract.--Analysis techniques, typically used in image processing, are well-suited to terrain stratification. Color composite techniques are used to create a single composite image that combines elevation, slope and aspect data for the area of a 22,000 acre wildfire in the San Mateo Mountains of New Mexico. Cluster images are then produced demonstrating the technique's utility for determining areas of homogeneous combination of terrain components.

INTRODUCTION

The sky islands of the Southwest are distinguishable from the surrounding landscape because of their topographic characteristics. These islands are isolated regions of great relief, steep slopes and high elevation. These same topographic characteristics are spatial determinants of many ecosystem phenomena, including the spatial behavior of wildfire. For example, fuel types, configurations, rates of accumulation, and fire behavior can all be influenced by topographic variables such as slope, elevation, and aspect (Burgan and Shasby 1984).

In analyzing the role of topography in wildfire research, it is necessary to identify areas that have similar combinations of topographic variables. This spatial information can help us recognize areas that are prone to certain fire behaviors, particular fuels configurations, or unique energy balance conditions, as a result of topography. If we had perfect information about the role of topographic parameters, it would be possible to use GIS to identify the areas which manifest the terrain of interest. However, in many applications we have insufficient understanding of the complexities of ecosystem functions to determine accurately the topographic characteristics associated with a given phenomenon. In order to incorporate terrain data when we have an imperfect understanding, it is necessary to develop automated techniques that can help us stratify terrain in a repeatable manner, which can also be applied at broad scales.

We demonstrate an automated technique, using digital image processing, that stratifies terrain into homogenous combinations of elevation, slope, and aspect. We use a false color combination protocol to first combine elevation, slope, and aspect data into a single composite image (see Figure 1). This image is then used as input for a clustering technique, which identifies homogeneous terrain. The analyst controls the clustering algorithm, thereby permitting the identification of a number of clusters that the analyst believes is most informative for the research in question (see Figure 2).

TECHNIQUES

Constructing the Digital Elevation Mosaic

A mosaic of 30 Meter USGS Digital Elevation Models (DEMs) was constructed for the area encompassing the Coffee Pot Fire in New Mexico in 1994. Missing elevation values that appeared along the edge of the source DEMs were replaced with averages of the neighboring pixels. A rectangular subscene of the mosaic was extracted from the concatenated DEM. Several peaks at the center of the DEM reach over 3200 meters. Lower portions of the mosaic lie around the 1800 meter level.

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[2] *Department of Geography and Regional Development, [2] University of Arizona, Tucson AZ.*

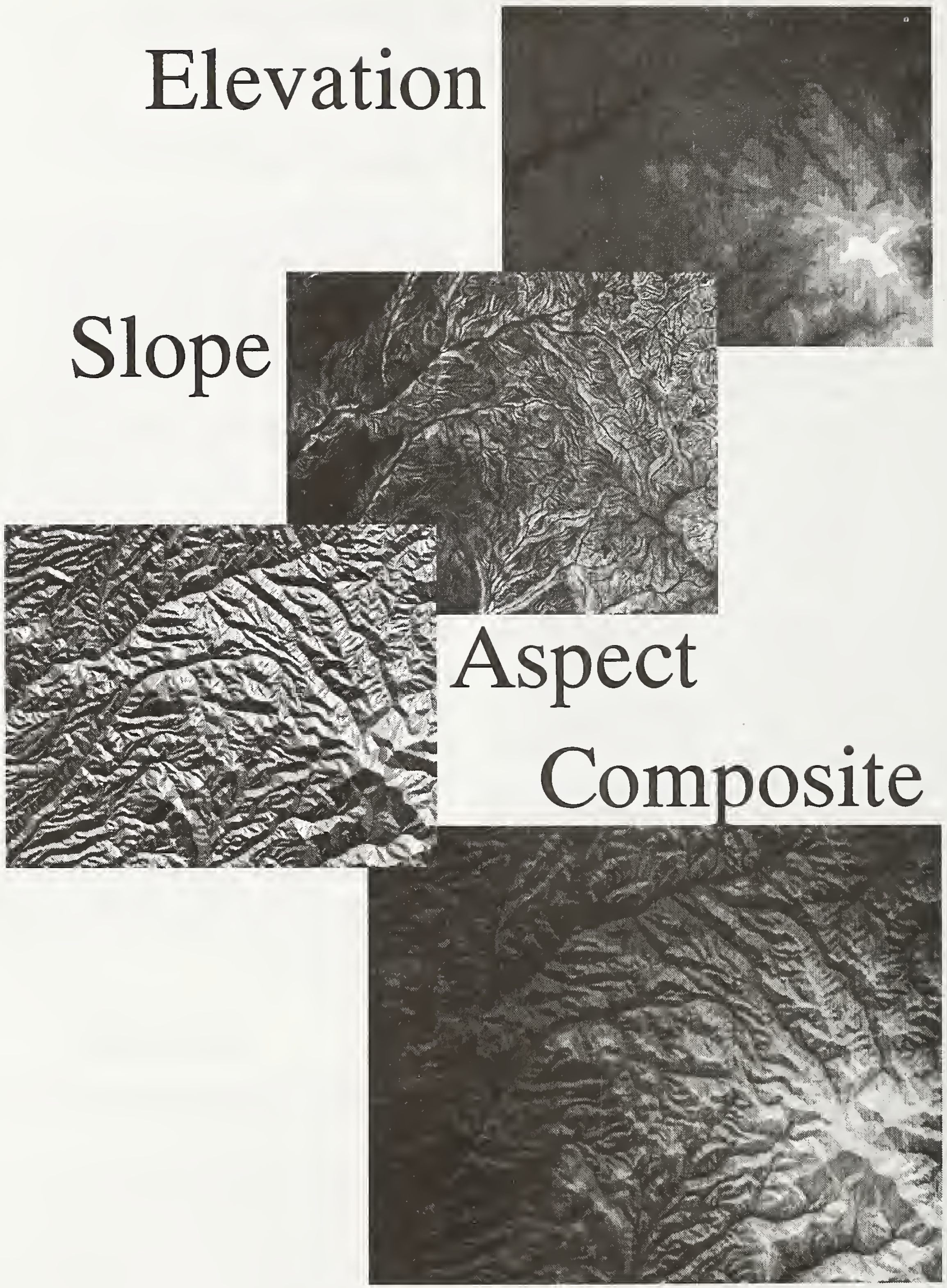

Figure 1. Three terrain images were used to create a composite image. Elevation was assigned the color green, slope was assigned blue, and aspect assigned red. [Colors used on original poster could not be reproduced here.] The brightness of the colors is a function of the value of each pixel in the input images. The composite image contains information in the three input images (easier to note in a color image). Elevation is visible because higher areas are brighter, aspect is visible as a false shading effect, and areas of low slope (ridge lines and drainage bottoms), are visible as linear features.

Figure 2. The composite image contains the sum of the spatial information in the three input images. Clustering was used to produce the three other images.

Producing the Slope, Aspect, and Elevation Images

The DEM mosaic was used to produce elevation, slope, and aspect images. The elevation image is simply a visual display of the elevation value for each pixel in the DEM, (see elevation image in Figure 1). Slope and aspect values were computed for each pixel from the information in each three-by-three pixel window in the DEM. This creates three spatially co-registered terrain images. The slope image shows areas of low slope (drainage bottoms, and ridge tops) as darker linear features, (see slope image in Figure 1).

Aspect statistics require special consideration because of their circular nature. Specifically 0 and 360 degrees both represent North. Unlike elevation and slope, which are linear, aspect measures are circular. We classified aspect into a 'rose' diagram of eight classes, each class a 45-degree arc. These aspect classes correspond to a combination of solar incidence and ambient air temperature while in direct sun, (see aspect image in Figure 1), (Gaile and Burt 1980). Therefore the northern most directions had the lowest values, while the southernmost directions had the highest values. These classes can be associated with differential fuel moisture and accumulation rates.

Creating a Composite Image

False-color composite images are typically used to show merged information from three spectral bands. A spectral band is assigned either red, green, or blue in the composite. A high value in any one band is shown by the intensity of the corresponding assigned color. We have used this protocol to assign colors to the values in the three input terrain images. Slope was assigned blue, elevation was assigned green, and aspect was assigned red. The false-color composite of these three images thus represents the aggregation of all information available in the three classified terrain images, (see composite image in Figure 1).

Clustering the Composite Image

Clustering of multi-spectral data is a common means of identifying dominant spectral response patterns (Lillesand 1994). Similarly, the composite terrain image, shown in both figures can be clustered into homogeneous combinations of slope, elevation, and aspect. A three-dimensional histogram peak technique was used, which detects the combinations of slope, elevation and aspect that occur most frequently, then assigns pixels to the nearest peak frequency. This technique is often used for unsupervised classification of false color composites. (Jensen 1996).

The number of clusters can be adjusted by the analyst. Originally the false color composite was clustered with all clusters retained. A histogram showing cluster modes was then used to aggregate similar cluster groups. The false-color composite was then re-clustered using the break points of these new groups. Remaining points are placed into the nearest cluster. Figure 2, for example, shows 8, 12, and 15 cluster images. In each of these images the clusters represent terrain with similar slope, elevation and aspect.

CONCLUSION

Our continuing wildfire research suggests terrain is a significant factor guiding the spatial patterns of fire effects. We have successfully incorporated the composite image presented here with satellite data, to improve the accuracy of "potential fire induced vegetation mortality" images, (Medler and Yool, Forthcoming). These potential fire images were produced from pre-fire imagery with and without the addition of the terrain composite image, and the results were tested against the burn patterns of the Coffee Pot Fire. The addition of the terrain composite image improved classification accuracy by over 30 percent.

Complex arrangements of elevation, slope, aspect, affect fire behavior, intensity, and severity. Forest fuel types, their arrangement, and rates of fuel accumulation are linked to terrain. The automated system we have described here offers researchers the means to stratify complex associations of terrain variables.

Computer-assisted terrain stratification can aid in identifying the dominant *spatial response patterns* in complex data without prior knowledge. It offers the opportunity to investigate terrain complexity, scale-dependence, and fire effects by manipulating the number of clusters or grain size. Thereby, we can analyze the effects of fire's spatial patterns on ecological processes (Turner 1994). Use of image processing techniques for terrain stratification may thus facilitate the capture of biophysical complexities that, in the past, were unavailable for such analysis. This

type of cluster image can also be easily draped over a three dimensional image of an area. This can facilitate easy visualization of the spatial arrangement of these terrain classes. Many studies in biogeography could benefit from easier and more intuitive techniques for the visualization of the spatial complexity of the terrain components that are determinants of many biophysical phenomena.

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Effects of Fire Frequency on Nutrient Budgets of Grasslands in Southeastern Arizona[1]

Thomas H. Biggs[2], Robert H. Webb[3], and Jay Quade[2]

Throughout the southwestern United States, vegetation in what historically was grassland has changed to a mixture of trees and shrubs with scattered perennial grasses. Livestock grazing and decreased fire frequency are considered the major causes of this conversion. Current strategies for ecosystem management are based on a slowing or outright reversal of this trend. The availability and amount of soil nutrients influence the relative success of plants, but the effects of fire frequency on soil nutrients is unknown for desert grasslands. Fires liberate bio-available phosphorus (PO_4-P), soluble nitrogen (NO_3), and carbon from above ground biomass, but considerable quantities of these nutrients would be volatilized, lost in aerosols, and removed in surface runoff. Our research has concentrated on the effects of fire on a soil at Fort Huachuca Military Reservation, where wildfire frequency history is known from 1973 to present, detailed soils mapping has been done, and no significant cattle grazing has occurred in recent decades. Subplots on this soil represent fire frequencies of no burns, 2 fires per decade, and 5 fires per decade. The "no burn" plot has abundant mature mesquite trees, whereas the burned plots are open desert grassland with scattered (and typically small) mesquite trees. Our results (Table 1) indicate that (1) fire frequency has altered the basic geochemistry and nutrient availabilities of the soil, and (2) burning is advantageous for preservation or restoration of grasslands, but too much burning can be detrimental to the ecosystem. Increasing fire frequency causes a decrease in soluble nitrate (NO_3), although most of this nutrient is sequestered in mesquite on the "no burn" plot. Total organic carbon, total nitrogen, and plant-available phosphorus (PO_4-P) show significant increases on the "infrequently burned" plot compared to the "no burn" site, whereas the "frequently burned" plot showed dramatic decreases in these nutrients. Soil pH increases with burning frequency, and cation exchange capacity, a key component of soil fertility, is 50% greater on burned plots compared to the "no burn" plot. Total living grass biomass is greater on the two burned plots, but variances are not significantly different for shrubs, forbs, and litter between plots. Root biomass is significantly lower and spatially less variable on the "frequently burned" plot, indicating decreased growth rates.

Table 1. Results of soil nutrient studies at three sites of different fire frequency on Fort Huachuca Military Reservation. The soil is a non-gravelly sandy loam of the Gardencan-Lanque complex. n= 121 samples, except Total N (10) and pH (20) / grid.

Grid	Fire frequency (# / decade)	pH	Total N(%)	Soluble N (mg/g)	Organic C(%)	PO_4 - P (mg/kg)
1	0	5.57	0.10	7.52	1.01	15.50
		(0.48)	(0.02)	(25.68)	(0.37)	(8.11)
2	2	5.75	0.11	3.71	1.18	24.86
		(0.34)	(0.03)	(3.77)	(0.44)	(7.52)
3	5	6.24	0.09	2.04	0.87	11.09
		(0.22)	(0.02)	(1.62)	(0.28)	(4.07)
Significance		*^	^	~*^	~*^	~*^

*[Preliminary t-test results: ~ indicates significant difference between grids 1 &2; * indicates significant difference between grids 1 & 3; ^ indicates significant difference between grids 2 and 3. All results are at the 90% confidence level.]*

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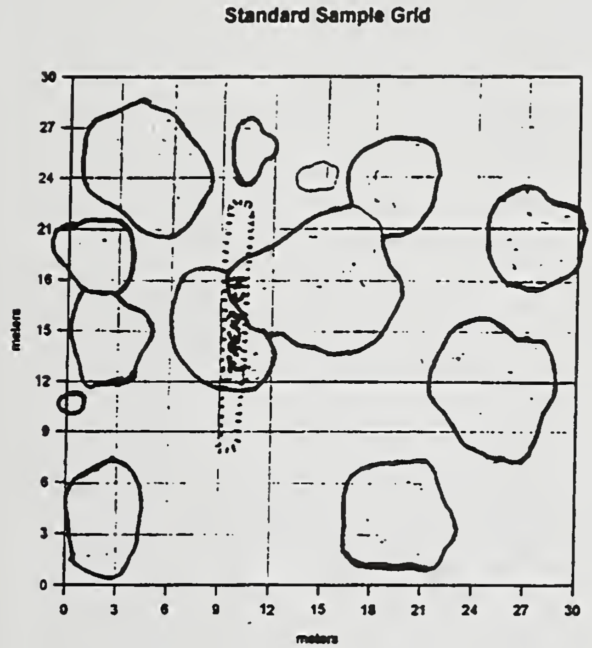
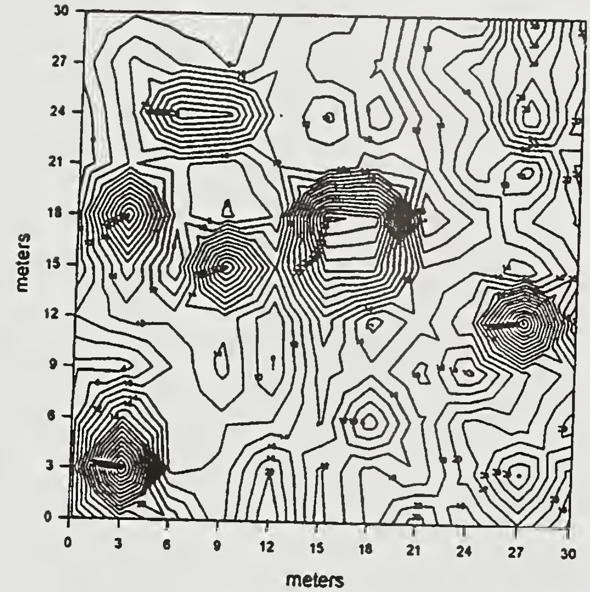
DISTRIBUTION OF MESQUITE CANOPIES
"NO BURNS" PLOT (GRID #1)

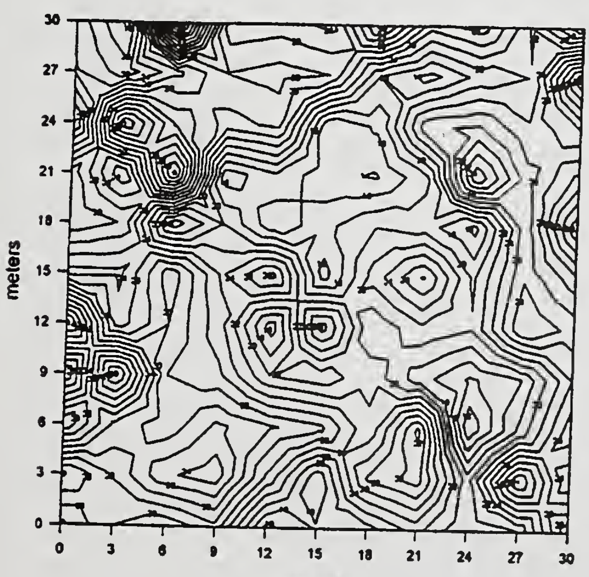


SOIL PHOSPHORUS (PO4-P):
"NO BURNS" PLOT (GRID #1)



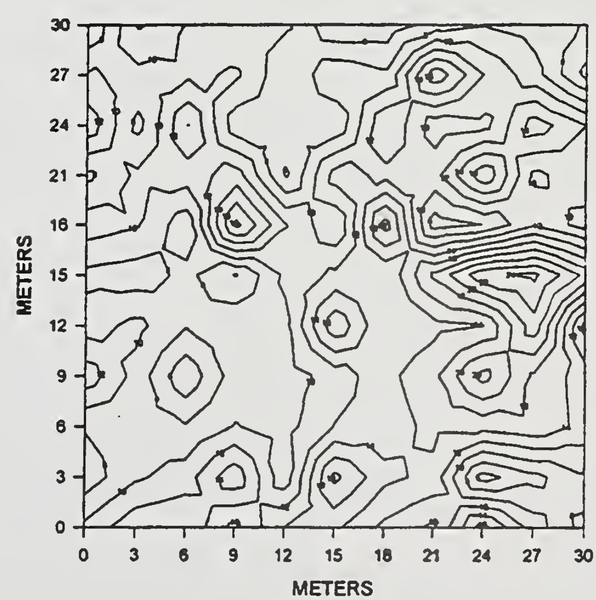
Contour interval: 2 mg/kg PO4-P

SOIL PHOSPHORUS (PO4-P)
"INFREQUENT BURNS" PLOT (GRID #2)



Contour interval: 2 mg/kg PO4-P

SOIL PHOSPHORUS (PO4-P)
"FREQUENT BURNS" PLOT (GRID #3)



Contour interval: 2 mg/kg PO4-P

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